Carbon sequestration and rangelands: A synthesis of land management and precipitation effects

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Abstract: Management of rangelands can aid in the mitigation of rising atmospheric carbon dioxide concentrations via carbon storage in biomass and soil organic matter, a process termed carbon sequestration. Here we provide a review of current knowledge on the effects of land management practices (grazing, nitrogen inputs, and restoration) and precipitation on carbon sequestration in rangelands. Although there was no statistical relationship between change in soil carbon with longevity of the grazing management practice in native rangelands of the North American Great Plains, the general trend seems to suggest a decrease in carbon sequestration with longevity of the grazing management practice across stocking rates. The relationship of carbon sequestration to mean annual precipitation is negative for both the 0 to 10 cm (0 to 3.9 in) and 0 to 30 cm (0 to 11.8 in) soil depths across stocking rates. The threshold from positive to negative carbon change occurs at approximately 440 mm (17.3 in) of precipitation for the 0 to 10 cm soil depth and at 600 mm (23.6 in) for the 0 to 30 cm soil depth. We acknowledge that largely unexplored is the arena of management-environment interactions needed to increase our understanding of climate-plant-soil-microbial interactions as factors affecting nutrient cycling. Continued refinement of estimates of terrestrial carbon storage in rangelands will assist in the development of greenhouse gas emissions and carbon credit marketing policies, as well as potentially modifying government natural resource conservation programs to emphasize land management practices that increase carbon sequestration.

Key words: grazing—legumes—nitrogen inputs—precipitation gradient—soil organic carbon (SOC)

Atmospheric carbon dioxide concentrations have varied historically, but they have substantially increased from 270 μmol mol^{-1} in 1870 prior to the Industrial Revolution to current levels greater than 365 μmol mol^{-1} (Morgan et al. 2004a). Land management practices, however, offer opportunities to mitigate the rise in atmospheric CO_2 concentration through sequestration of this additional carbon via storage in plant biomass and soil organic matter in a process termed terrestrial C sequestration (Izaurralde et al. 2001). Carbon sequestration also provides associated ecosystem co-benefits such as increased soil water holding capacity, better soil structure, improved soil quality and nutrient cycling, and reduced soil erosion.

Rangelands have a large potential to sequester C because they occupy about half of the world's land area and store greater than 10% of terrestrial biomass C and 10 to 30% of global soil organic carbon (SOC) (Schlesinger 1997; Scurlock and Hall 1998). It is estimated that rangelands globally sequester C in soil at a rate of 0.5 Pg C yr^{-1} (Schlesinger 1997; Scurlock and Hall 1998). Although soil C sequestration rates are low on rangelands relative to those reported for croplands and improved pastures, increases in terrestrial C on rangelands resulting from management account for a significant amount of C sequestration given the large geographical area of this land resource and generally require minimal inputs. This implies that modest changes in C storage in rangeland ecosystems have the potential to modify the global C cycle and indirectly influence climate (Schimel et al. 1990; Ojima et al. 1993; Conant et al. 2001). Despite its significance, our understanding of land use effects on the storage of C in rangelands remains limited (Schuman et al. 2001; Reeder and Schuman 2002). In addition, research addressing C sequestration in rangelands is still in its infancy when compared to cropland and forestry.

Soil C sequestration on rangelands is influenced by biome, climate (Conant et al. 2001), management practices, and environmental factors (Jones and Donnelly 2004). Even though C dynamics on rangelands involve complex interactions involving climate, soils, plant communities and management, we currently only have a rudimentary knowledge of these interactions as controlling drivers influencing soil C sequestration (Schuman et al. 2001). There is, however, emerging evidence that the relative contribution of management practices can be lower compared to climatic drivers (e.g., Schuman et al. 2005). Nevertheless, there remains a paucity of basic ecological information that is needed to improve understanding of why, when and where rangeland ecosystems function as C sinks or sources.

Here we provide a review of current knowledge on the effects of land management practices (grazing, N inputs via fertilization and interseeding of N-fixing legumes, and restoration of degraded lands) on C sequestration in rangelands. Because of the limited information on C sequestration in dry lands including deserts and desert grasslands (e.g., Gardner 1950; Rasmussen and Brotherson 1986; Lal 2004a; Vagen et al. 2005), this paper primarily addresses semi-arid (250 to 500 mm annual precipitation) and mesic (500 to 1000 mm) rangeland ecosystems where the vast majority of research has been conducted. These data are synthesized to determine relationships of changes in soil C to length of time a rangeland management practice has been in place and to gradients in mean annual precipitation as C sequestration rates increase from arid (0.02 to 0.08 Mg C ha^{-1} yr^{-1}) to semi-arid (0.03 to 0.12 Mg C ha^{-1} yr^{-1}) to semi-humid and sub-humid (0.08 to 0.20 Mg C ha^{-1} yr^{-1}) environments (Lal 2000). Differences in above-ground versus below-ground constraints on plant growth (Burke et al. 1998), root/shoot ratios

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Table 1
Management effects on soil organic carbon sequestration rates of rangelands across ecosystems.

Management practice/ecosystem	Location	SOC sequestration	Citation
Grazing			
Shortgrass prairie	Colorado	0.12 Mg C ha^{-1} yr^{-1}	Derner et al. 1997
	Colorado	0.07 Mg C ha^{-1} yr^{-1}	Reeder and Schuman 2002
Northern mixed-grass prairie	Wyoming	0.30 Mg C ha^{-1} yr^{-1}	Schuman et al. 1999
	North Dakota	0.29 Mg C ha^{-1} yr^{-1}	Frank 2004
Southern mixed-grass prairie	Oklahoma	No change	Fuhlendorf et al. 2002
Nitrogen fertilization			
Tallgrass prairie	Kansas	1.6 Mg C ha^{-1} yr^{-1}	Rice 2000
Conservation Reserve Program	Wyoming	0.41 to 1.16 Mg C ha^{-1} yr^{-1}	Reeder et al.1998
	Saskatchewan	Increases of 5.4 to 9.3 Mg C ha^{-1} yr^{-1}	Nyborg et al. 1994
Legume interseeding			
Northern mixed-grass prairie	South Dakota	0.33 to 1.56 Mg C ha^{-1} yr^{-1}	Mortenson et al. 2004
Restoration			
Southern mixed-grass prairie	Oklahoma	No difference 0 to 10 cm with moderate grazing, but 65% decrease with heavy grazing	Fuhlendorf et al. 2002
	Sudan	Restored soil C to 80% of native rangeland in 100 yr	Olsson and Ardö 2002
Semiarid savanna	Argentina	466% increase	Abril and Bucher 2001
Tallgrass prairie	Texas	Average 0.45 Mg C ha^{-1}, estimated 100 years to achieve native rangeland level	Potter et al. 1999
Conservation Reserve Program	Texas, Kansas, and Nebraska	0.8 to 1.1 Mg C ha^{-1} yr^{-1}	Gebhart et al. 1994
	Texas to North Dakota	Average 0.9 Mg C ha^{-1} yr^{-1}	Follett et al. 2001
Mined lands	Wyoming	400% increase over 30 years	Stahl et al. 2003

Note: Data from Schuman and Derner 2004; Follett and Schuman 2005.

(Sims et al. 1978; Derner et al. 2006), root C/SOC ratios (Derner et al. 2006), and productivity potential (Knapp and Smith 2001) have been previously demonstrated across precipitation gradients.

Land Management Effects on Carbon Sequestration

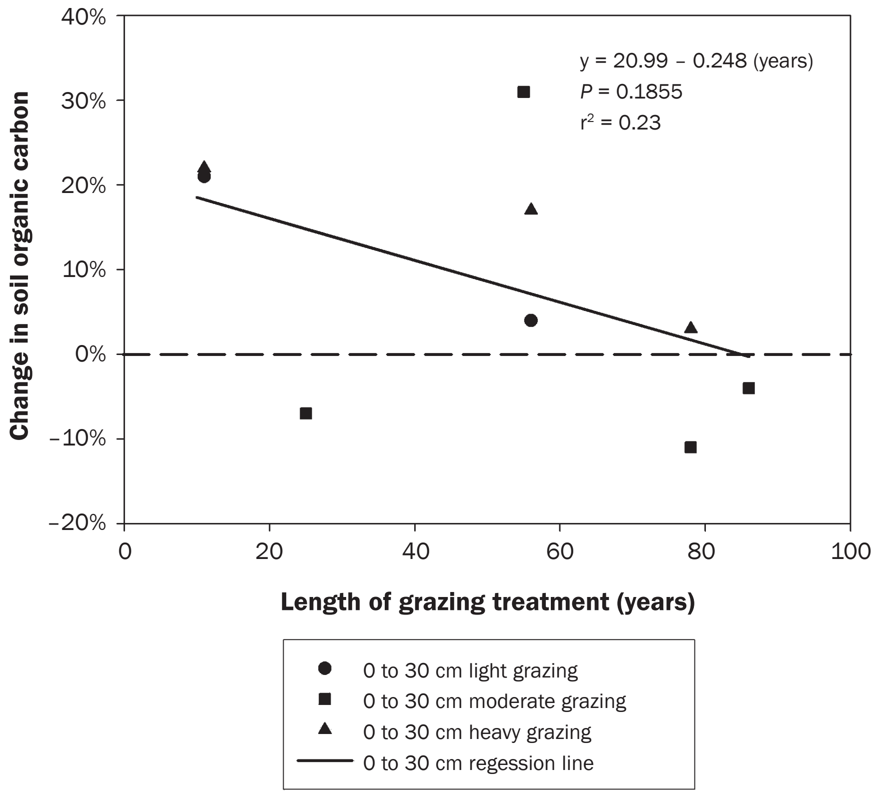
Grazing. Grazing facilitates the physical breakdown, soil incorporation and rate of decomposition of residual plant material (Naeth et al. 1991; Shariff et al. 1994; Schuman et al. 1999). Grazing intensity and frequency are thought to cause the primary effects on C storage across rangelands (Bruce et al. 1999), although these effects are often inconsistent and difficult to predict (Milchunas and Lauenroth 1993; Schuman et al. 2001; Reeder and Schuman 2002).

Grazing-induced changes in plant community composition are likely responsible for many of the changes in C sequestration observed with stocking rates. For example, moderate and heavy stocking rates during the grazing season employed in a shortgrass steppe and a northern mixed-grass prairie modified the plant community composition by reducing the proportion of cool-season (C3) perennial grasses while increasing the predominant warm season (C4) perennial grass, blue grama (Frank et al. 1995; Schuman et al. 1999; Derner et al. 2006). This change in plant community composition reduces the production potential of these rangelands by up to one-third (Schuman et al. 1999) but increases SOC because of the greater transfer of C to belowground plant parts in blue grama (Coupland and Van Dyne 1979). Grazing at moderate and heavy stocking rates in a shortgrass steppe increased SOC in the surface 30 cm (11.9 in) compared to adjacent nongrazed exclosures (Derner et al. 1997, 2006; Reeder and Schuman 2002; Reeder et al. 2004), resulting in estimated C sequestration rates of 0.12 Mg C ha^{-1} yr^{-1} (108.78 lbs ac^{-1} yr^{-1})for the moderate stocking rate and 0.07 Mg C ha^{-1} yr^{-1} (63.46 lbs ac^{-1} yr^{-1}) for the heavy stocking rate (table 1).

Also, grazing at light or heavy stocking rates in northern mixed-grass prairie increased SOC in the surface 30 cm of the soil compared to nongrazed exclosures (Schuman et al. 1999), resulting in a C sequestration rate of 0.30 Mg C ha^{-1} yr^{-1} (271.96 lbs ac^{-1} yr^{-1}) (table 1). Carbon sequestration rates, calculated using CO_2 flux data, on a northern mixed-grass prairie were similar at 0.29 Mg C ha^{-1} yr^{-1} (262.89 lbs ac^{-1} yr^{-1})(Frank 2004). Long-term (81 years) moderate and heavy grazing in northern mixed-prairie increased soil C by 19% and 34%, respectively, in the surface 5 cm (1.9 in) of soil compared to nongrazed exclosures (Wienhold et al. 2001). Grazing strategy did not affect C sequestration in a northern mixed-grass prairie as no differences were evident among short-duration rotational grazing, rotationally-deferred grazing, and continuous season-long grazing at heavy stocking rates (Manley et al. 1995).

There is a paucity of information addressing the interactions of management and environment on C sequestration. One recent study, however, sheds considerable light into the complexity of these interactions and the limitations in extending data from relatively short-term investigations to long-term predictions. Severe drought and heavy grazing can result in significant losses of SOC that was previously stored during normal to above-normal production years in northern mixed-grass prairie (Schuman et al. 2005). Loss of SOC during these drought years concurs with CO_2 flux data results during the same time period in a nearby shortgrass prairie (Morgan et al. 2004b). Microbial community shifts and activity were also observed in the heavy stocking rate of the northern mixed-grass prairie that exhibited significant losses in soil C during the severe drought period (Ingram et al. 2004, Stahl et al. 2004). These data suggest that additional information is needed to assess the importance and processes associated with dramatic fluctuations in climatic factors, such as drought, that may induce losses of C in rangeland ecosystems (Breshears and Allen 2002). Drought may change rangelands from sinks to sources of atmospheric CO_2 because limiting soil water proportionally affects photosynthetic rates more than total respiration (Balogh et al. 2005.)

Figure 1
Percent change (grazed vs. nongrazed) in soil organic carbon with respect to length of grazing treatment in the North American Great Plains.



Note: Data from Frank et al. 1995; Schuman et al. 1999; Reeder and Schuman 2002; Derner et al. 2006.

Nitrogen Inputs. Many rangelands are N deficient and have been shown to exhibit increased production (Samuel and Hart 1998; Berg and Sims 2000) and water-use-efficiency in response to N addition (Power and Alessi 1971). Addition of N fertilizer increased soil C in a tallgrass prairie (Rice 2000), in CRP lands (Reeder et al. 1998), in rangelands in Saskatchewan, (Nyborg et al. 1994) and in Alberta (Malhi et al. 1991) (table 1).There are, however, substantial C emissions associated with the production, packaging, storing and distribution of N fertilizers (Lal 2004b). Changes in SOC on "WW-Spar" Old World bluestem (*Bothriochloa ischaemum* L.) pastures in Oklahoma after five years of annual N fertilizer applications were greater for the intermediate application rate (68 kg N ha^{-1} yr^{-1}) compared to lower (34 kg N ha^{-1} yr^{-1}) and higher (102 kg N ha^{-1} yr^{-1}) application rates (Berg and Sims 2000). Application of other nutrients, where they are deficient, can also enhance SOC storage (Nyborg et al. 1998, 1999; Conant et al. 2001). The benefits of increased SOC sequestration with N-fertilization are offset, however, by emissions of CO_2 and N_2O in the fertilizer process (Schlesinger 1999, 2000), as well as the enhancement of NO_x (nitric oxide plus nitrous oxide) emissions and reduction of CH_4 uptake in soils (Mosier et al. 1998).

The introduction of N-fixing legumes into rangelands has been the subject of research for decades to provide an alternative to N-fertilization (Tesar and Jakobs 1972; Heinrichs 1975; Kruger and Vigil 1979; Berdahl et al. 1989). For example, interseeding yellow-flowered alfalfa (*Medicago sativa* ssp. *falcata*) into northern mixed-grass prairie increased SOC by 4% to 17% across three interseeding dates (Mortenson et al. 2004), resulting in C sequestration rates of 1.56, 0.65, and 0.33 Mg C ha^{-1} yr^{-1}, for 3-, 14-, and 36-year post-interseeding, respectively (table 1). N fixation by the yellow-flowering alfalfa also significantly increased soil total N, aboveground production, and N forage quality (Mortenson et al. 2004, 2005). This increase in production accounts for the enhanced SOC storage and does not represent any "C costs" in the production of N. An initial report by Schuman et al. (2004)

indicated no increase in the emission of the greenhouse gas nitrous oxide from rangeland soils on which yellow-flowered alfalfa had been interseeded.

Restoration of Degraded Lands

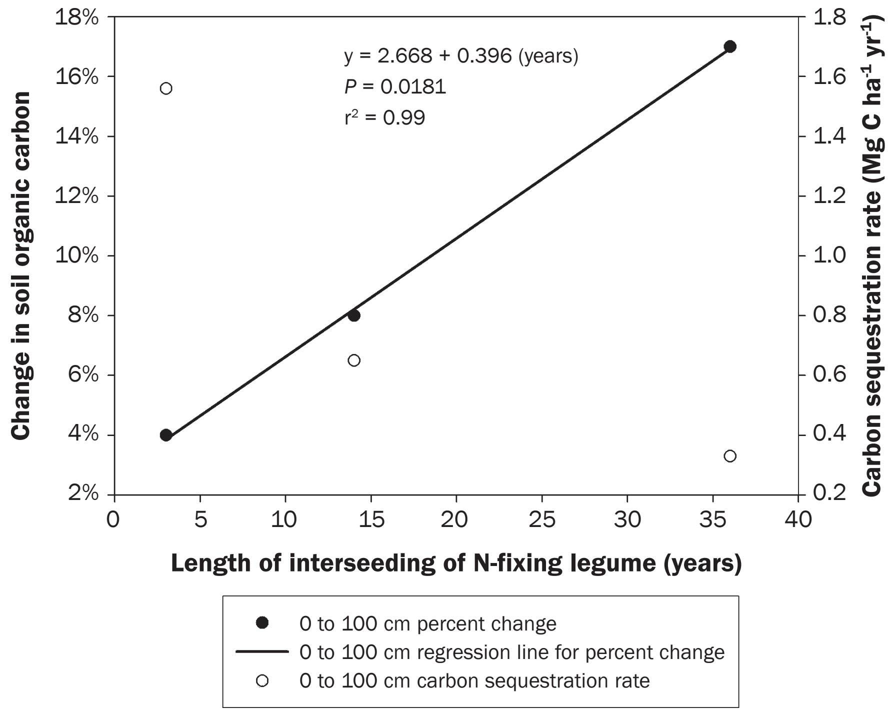
Cultivated Lands. Cultivation of rangeland soils dramatically reduces SOC (Haas et al. 1957). For example, shortgrass steppe soils cultivated for 60 years had 62% less SOC in the upper 15 cm (5.9 in) of the soil profile compared to native rangeland (Bowman et al. 1990). Abandoning cultivation of the shortgrass steppe soils, without re-establishing permanent vegetation, has increased SOC by 20% in the surface 10 cm (3.9 in) over 50 years (Burke et al. 1995); yet these soils contained only 67% of the SOC found in native shortgrass steppe, suggesting that there is a considerable capacity to sequester C in these degraded soils. This capacity may be tempered, however, by limitations associated with the loss of soil productivity due to historic erosional losses of topsoil while these lands were under cultivation.

Restoring permanent vegetation on formerly cultivated lands offers opportunities to store substantial amounts of SOC. For example, Post and Kwon (2000) report an average soil C sequestration rate for grassland establishment on formerly cultivated lands of 0.33 Mg C ha^{-1} yr^{-1} (299.15 lbs ac^{-1} yr^{-1}), although they acknowledge there is considerable variation. Grazing can alter the recovery of these restored sites and affect C sequestration as heavy grazing can reduce the rate of C accumulation (Fuhlendorf et al. 2002). Predicted recovery of soil C on cultivated fields to pre-cultivation levels without restoration efforts is 230 years (Knops and Tilman 2000). Model estimates indicate that conversion of marginal agricultural lands to rangeland in Sudan would restore SOC levels to 80% of those found in native savannas in 100 years (Olsson and Ardö 2002). Potter et al. (1999) estimated restoration efforts in a tallgrass prairie would require 158 years for the restored site to have a similar C pool as native prairie.

Carbon sequestration rate estimates for restored lands range from 0.28 Mg C ha^{-1} yr^{-1} (253.83 lbs ac^{-1} yr^{-1}) in the surface 20 cm (7.9 in) on highly restored sites in a semi-arid savanna in the western Chaco of Argentina (Abril and Bucher 2001) to 0.90 Mg C ha^{-1} yr^{-1} for Conservation Reserve Program lands from Texas to North Dakota

Figure 2

Percent change (interseeded vs. noninterseeded) in soil organic carbon and carbon sequestration rate with respect to length of interseeding of N-fixing legume in a northern mixed-grass prairie.



Note: Data from Mortenson et al. 2004.

Figure 3

Percent change (restored vs. nonrestored) in soil organic carbon with respect to length of restoration of formerly cultivated lands in North American Great Plains.

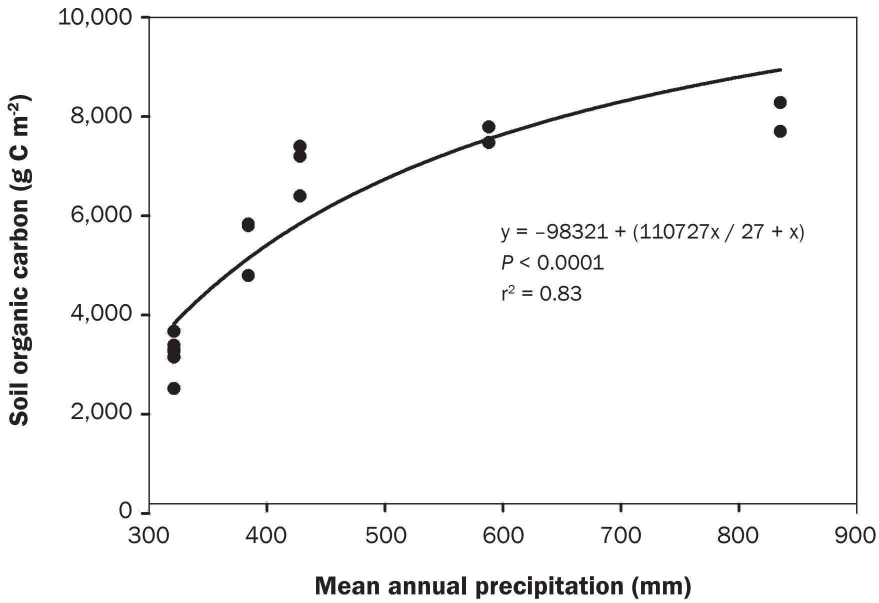
Note: Data from Gebhart et al. 1994; Burke et al. 1995; Reeder et al. 1998; Karlen et al. 1999; Potter et al. 1999.

(Follett et al. 2001) (table 1). Similarly, C sequestration rates in the 0 to 40 cm (0 to 15.7 in) and 0 to 300 cm (0 to 118.1 in) soil depths were estimated to be 0.8 to 1.1 Mg C ha^{-1} yr^{-1} (725.22 to 997.17 lbs ac^{-1} yr^{-1}) for Conservation Reserve Program lands in Texas, Kansas and Nebraska (Gebhart et al. 1994). Restored tallgrass prairie in Texas was estimated to have a C sequestration rate of 0.45 Mg C ha^{-1} yr^{-1} (407.93 lbs ac^{-1} yr^{-1}) in the upper 60 cm (23.6 in) of the soil profile (Potter et al. 1999).

Mined Lands. Restoration of rangelands disturbed by surface mining represent a minor fraction of the land area in the United States. Only 2.3 M ha (5,683,423 ac) of land was permitted for surface coal mining in the United States between 1977 and 2001 (Galetovic 2005), but these lands have great potential for high rates of C sequestration because of the soil salvage process. Soil salvage generally results in the collection of the two surface soil horizons and in some cases stockpiling of this material for several years. This process results in the dilution of the SOC pool by mixing of the soil organic matter (SOM) rich surface horizon with subsoil horizons that have lower amounts of SOM (Woods and Schuman 1986; Schuman 2002). The resulting soil material has similar levels of SOM to those found in dryland cropland soils that were cultivated for > 50 years (Haas et al. 1957; Tiessen et al. 1982; Burke et al. 1989; Bowman et al. 1990); therefore, these soils have a SOC sequestration potential similar to marginal, highly erodible croplands that have been restored to grasslands. Stockpiling soil also greatly diminishes the quality of the soil through enhanced organic matter degradation/decomposition, loss of plant residue inputs and general loss of much of the microbial functions (Severson and Gough 1983; Harris et al. 1989, 1993). Alternative plant growth materials are also sometimes used in place of topsoil resulting in subsoil or mine spoil material with very low levels of SOC and limited nutrient cycling potential (Schuman and Taylor 1978; Woods and Schuman 1986). Reclaimed mine soils (0 to 15 cm depth) in Wyoming have exhibited increases in SOC of about 400% over a 30-year period (Stahl et al. 2003) (table 1). These authors hypothesized that the decomposition rates are reduced in reclaimed mine soils due to low microbial activity, thereby accounting for the larger than expected increase in C.

Figure 4

Soil organic carbon pools to 30 cm depth across a mean annual precipitation gradient in the North American Great Plains.



Note: Data from Frank et al. 1995; Schuman et al. 1999; Reeder et al. 2004; Derner et al. 2006.

Synthesis of Land Management Effects on Carbon Sequestration in Rangelands

Length of Practice. Although there was no statistical relationship between change in SOC with longevity of the grazing management practice in native rangelands of the North American Great Plains, the general trend seems to suggest a decrease in C sequestration with longevity of the grazing management practice across stocking rates (figure 1). This trend in linear reduction in C sequestration of the top 30 cm (11.8 in) of the soil profile is consistent with the understanding that the ecosystem will reach a 'steady-state' and a change in management and/or inputs would be required to sequester additional C (Conant et al. 2001, 2003; Swift 2001).

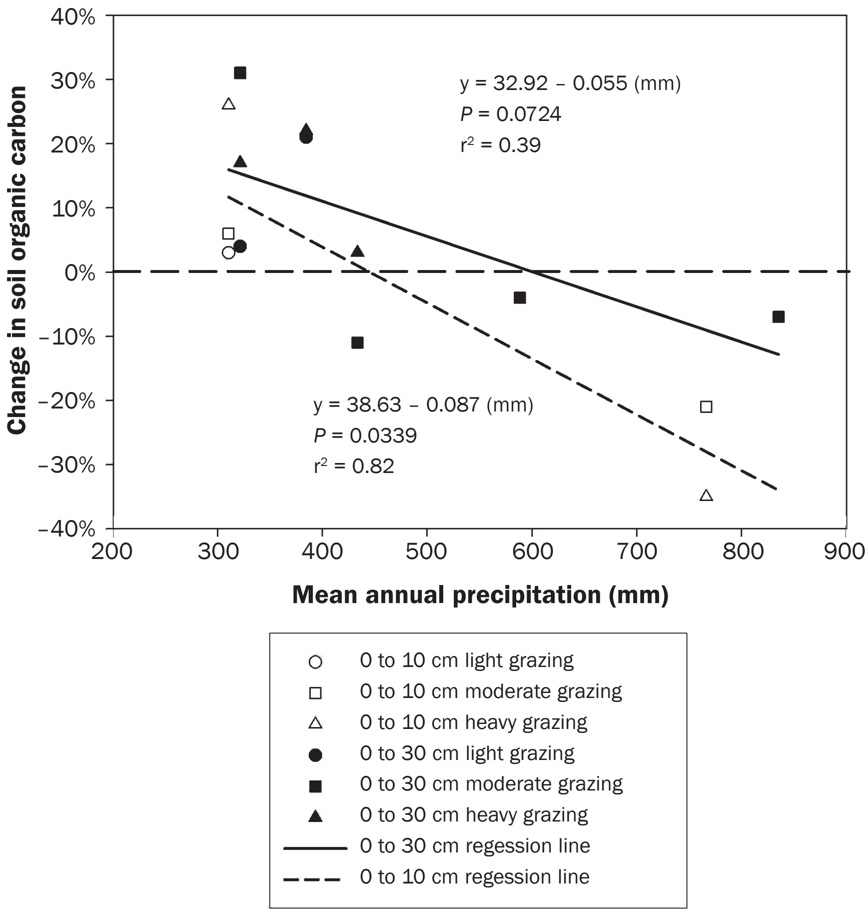
In contrast to grazing, the general relationship of change in SOC to longevity of the interseeding of a N-fixing legume is positive with time (figure 2). It should be noted, however, that this relationship is based on a single study. Nevertheless, interseeding of an N-fixing legume in rangelands likely has the potential to continue to increase inputs of C, and illustrates the importance of N in C sequestration. Management practices that maintain or slightly increase soil N, such as grazing (Johnston et al. 1971; Smoliak et al. 1972; Manley et al. 1995; Dormaar and Willms 1998; Burke et al. 1999; Schuman et al. 1999; Verchot et al. 2002; but see Frank et al. 1995; Frank and Evans 1997), will likely result in limited C sequestration. Management practices that reduce soil N will result in net C losses from the system.

The lack of a general relationship between C sequestration and longevity of restoration of perennial grass on formerly cultivated lands results from the high variability in changes in C reported for numerous studies where the length of restoration is less than 10 years (figure 3). This lack of a relationship is in contrast with increasing changes in SOC with length of restoration reported in central Texas by Potter et al. (1999) for the 0 to 60 cm (0 to 23.6 in) soil profile.

Mitigation of increasing atmospheric CO_2 is partially achievable through proper land management on rangelands. Additional data is needed to further develop the relationship between N-fixing legumes and length of treatment to increase our degree of confidence. Land management practices that increase soil N, such as interseeding of N-fixing legumes have the potential to continue sequestering C for longer time periods, and without the "C-costs" associated with the production of inorganic N fertilizers.

Precipitation Gradients. In rangelands of the North American Great Plains, SOC pools to a depth of 30 cm (11.8 in) have been shown to increase with increasing precipitation (figure 4). Pools of SOC are two to three

Figure 5
Percent change (grazed vs. nongrazed) in soil organic carbon with mean annual precipitation with grazing in North American Great Plains.



Note: 0 to 10 cm data from Smoliak et al. 1972 and Fuhlendorf et al. 2002; 0 to 30 cm data from Frank et al. 1995, Schuman et al. 1999, Reeder and Schuman 2002, Derner et al. 2006.

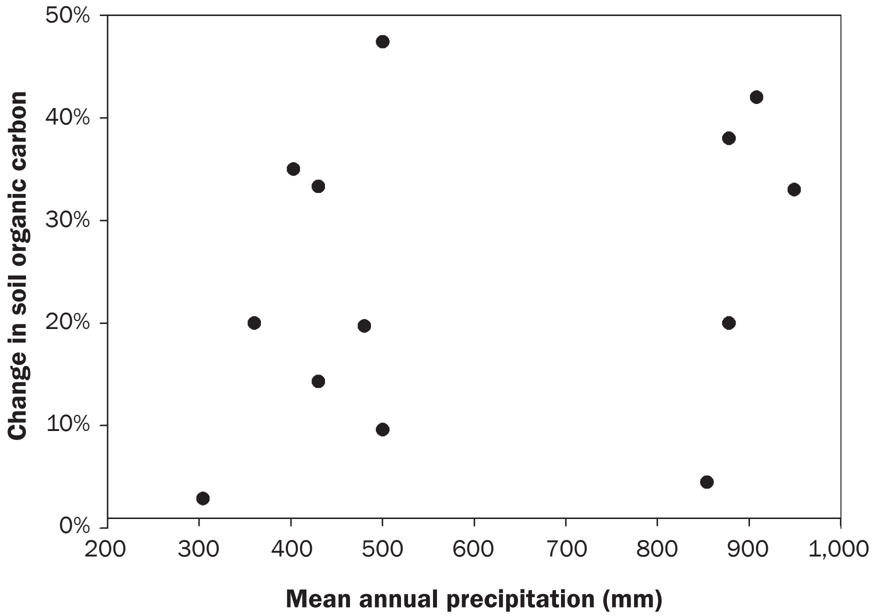
times lower in semi-arid than mesic rangelands (Derner et al. 2006). The same absolute change in SOC pools in semi-arid and mesic rangelands would result in a greater relative change in SOC pools for the semi-arid rangeland. Differential responses of SOC between semi-arid and mesic rangelands in the Great Plains to grazing are thought to be a result of 1) lower SOC pools, 2) greater root C/soil C ratios, and 3) a grazing-induced compositional shift to greater C4 dominance in the semi-arid shortgrass steppe (Derner et al. 2006). These authors hypothesized that the primary driver linking grazing-induced compositional changes in species composition to the C changes in shortgrass steppe was the magnitude and proportion of fine root mass in the upper soil profile (Derner et al. 2006).

The relationship of C sequestration to mean annual precipitation is negative for both the 0 to 10 (0 to 3.9 in) and 0 to 30 cm (0 to 11.8 in) soil depths across stocking rates (figure 5). The threshold from positive to negative C change occurs at approximately 440 mm (17.3 in) of precipitation for the 0 to 10 cm soil depth and at 600 mm (23.6 in) for the 0 to 30 cm soil depth. Above these threshold precipitation values, C sequestration does not increase and may actually decrease SOC. The observed relationship for the 0 to 10 cm soil depth along this precipitation gradient is in general agreement with Sims et al. (1978) and Sala et al. (1988) in identifying the 370 to 400 mm (14.6 to 15.8 in) precipitation range as the region in which a transition in aboveground ecosystem responses to grazing occurs, and this aboveground difference may also be manifested belowground. Not surprisingly, no differences for SOC were observed between nine long-term grazed and ungrazed (20 to 71 years) sites along a precipitation gradient of 330 to 480 mm (13.0 to 18.9 in) in Canada (Henderson et al. 2004). A potential mechanism responsible for the transition in C sequestration from semi-arid to more mesic environments may be the control of precipitation on N turnover and in interaction with N availability in controlling C gain (Austin and Sala 2002). The reduction in C sequestration in relatively wet environments can be explained by greater microbial biomass C and N and labile organic matter pools which increase nutrient cycling (Zak et al. 1994).

A relationship of C sequestration to mean annual precipitation for interseeding N-legumes cannot be assessed at this time because of a lack of data. Several interseedings of yellow-flowering alfalfa across Wyoming and adjacent states in the past two years should provide the opportunity to develop such a relationship in coming years. In addition, there is also a lack of any general relationship of C sequestration to mean annual precipitation for restoration of perennial grass on formerly cultivated lands (figure 6). The high variability in change in SOC to mean annual precipitation may be, in part, because of differences in the length of time sites were cultivated prior to restoration efforts, intensity of cultivation efforts, and types of species used in the restoration efforts.

Constraints to Synthesis. We note that the high degree of variability in soils and vegetation at multiple spatial scales ranging from plant community interspaces (Hook et al. 1991; Vinton and Burke 1995; Derner et al. 1997) to the landscape (Burke et al. 1999) necessitates that researchers select appropriate and comparable sampling sites across treatments for initial and subsequent sample collection to facilitate field data estimates of C sequestration rates (Schuman and Derner 2004). Furthermore, the concentration of SOC near the soil surface in rangelands (Weaver et al. 1935; Gill et al. 1999) has resulted in many studies addressing only responses of the uppermost portion of the soil profile (Burke et al. 1999). Deeper depths have also been shown to sequester C (Schuman et al. 1999), and C turnover is influenced by decomposition rates of roots which decrease with increasing soil depth (Gill and Burke 2002).

Figure 6
Percent change (restored vs. nonrestored) in soil organic carbon change with mean annual precipitation with restoration of formerly cultivated lands in North American Great Plains.



Note: Data from Gebhart et al. 1994; Burke et al. 1995; Reeder et al. 1998; Karlen et al. 1999; Potter et al. 1999.

Summary and Conclusions

Future Research Arenas involving Carbon Sequestration. Research is needed across multiple locations addressing key ecological processes and mechanisms to determine the principal drivers affecting C sequestration. The continued development of sophisticated in situ and laboratory equipment to accurately detect small but ecologically-important changes in soil C and its components will open new horizons for future experimentation and verification of C change due to management options or climatic variances. Newly emergent fields of soil microbial ecology should provide additional insight into microbial function and processes that affect C sequestration under normal and the widely fluctuating precipitation patterns found in arid and semi-arid environments. There is a need to move from the basic approach of soils and soil ecology to a more fundamental and functional understanding of the processes and mechanisms that affect SOC dynamics and how they are influenced by land management, environment and their interaction. For example, management strategies may offer opportunities to enhance soil fungal activity and C storage (Bailey et al. 2002). Largely unexplored is the arena of management-environment interactions that will increase our understanding of climate-plant-soil-microbial interactions as control factors affecting nutrient cycling within the context of determining costs and benefits (i.e., risk assessment) associated with varying land management practices. Management-environment interaction investigations should emphasize greater recognition of complete greenhouse gas budgets, thereby determining C storage and sequestration, as well as greenhouse gas emissions.

Existing long-term grazing studies, initially set up to evaluate livestock performance and vegetation change, provide excellent field laboratories to overlay additional contemporary treatments (e.g., prescribed fire and results of global climate change such as increased temperatures, altered precipitation patterns, carbon dioxide enrichment, N fixation/deposition) to more fully understand how and which best management practices affect greenhouse gas budgets (C sequestration and global warming). These efforts can be conducted within the established scientific missions of the USDA Agricultural Research Service, Agricultural Experiment Stations, and the Long-Term Ecological Research Network. Of paramount interest is the causal role that grazing-mediated changes in vegetation composition, productivity and diversity have in the interpretation of SOC responses. These efforts should be conducted within the framework of ecological sites, with particular attention directed at those vegetation states which are 'at-risk' of transitioning to another vegetation state.

Policy Implications. Estimates of soil C storage and rates of C sequestration for rangelands are being used by scientists and policymakers to estimate the potential of these lands to help mitigate the elevated atmospheric levels of CO_2 (Lal et al. 2003). Considerable interest is being generated in terrestrial C storage and marketing of stored C (Williams et al. 2004). Economic benefits from C sequestration programs have the potential to significantly contribute to household economies (Olsson and Ardö 2002). Continued refinement of estimates of terrestrial C storage in rangelands will assist in the development of greenhouse gas emissions and C credit marketing policies.

Existing U.S. government policies related to private land management are increasingly conservation-oriented as evidenced by programs within the 2002 Farm Bill relevant for rangelands which include the (1) Conservation of Private Grazing Land Program, (2) Conservation Security Program, (3) Environmental Quality Incentives Program, (4) Grassland Reserve Program, (5) Wildlife Habitat Incentives Program, and (6) Wetlands Reserve Program. Collectively, these programs emphasize monetary incentives for private land managers to adopt/maintain conservation practices, but these programs may be modified to emphasize land management practices that increase C sequestration.

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