

The Potential of U.S. Grazing Lands to Sequester Carbon and Mitigate the Greenhouse Effect

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CHAPTER 11

The Dynamics of Soil Carbon in Rangelands

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Introduction

The terrestrial biosphere contains large reserves of carbon (C) — about 1500 Pg of C in the surface meter of soil (Batjes, 1996; Eswaran et al., 1995) and another 600 Pg of C in vegetation (Houghton, 1995; Schimel, 1995). Together, these pools contain three times as much C as the atmosphere and, consequently, a small change in C storage in plants or soils has important implications for atmospheric CO₂. This relationship has gained attention with the recognition that atmospheric CO₂ content is increasing and with the consequent reevaluation of ways to increase C storage in the biosphere and thereby reduce atmospheric CO₂ levels.

Grazing lands occupy about 3 billion ha of land worldwide, about twice the area devoted to cultivated agriculture (Buyanovsky and Wagner, 1998; Bronson et al., 1997). These grazing land soils contain large reserves of C, especially in temperate regions, where soil C in the surface meter accounts for 12.5 to 18.4 kg C/m² (Paustian et al., 1997). The U.S. has about 336 Mha of grazing lands, which include about 161 Mha of rangeland, most of which is in the Great Plains region of the country (Sobecki et al., Ch. 2).

Highest rates of C gain in rangeland soils occur early in soil formation. With time, rates of accumulation diminish as the soils approach a new equilibrium or steady-state level (Schlesinger, 1990, 1995; Chadwick et al., 1994). Computer simulation suggested that much of the C accumulation in a rangeland soil occurred within the first 5000 years of development (Parton et al., 1988). Consequently, most soils in U.S. rangelands may now be past the stage of rapid C accrual.

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But that does not mean the soil C is static. Changes in soil C may still occur in response to a wide range of management and environmental factors. Given the sheer size of the C pool in U.S. rangelands, we need to understand these potential changes to best manage these systems for C storage and continued productivity.

C dynamics of rangelands is a very complex issue which climate, soils, plant community, and management affect. Research has reported very mixed and inconsistent responses to these variables because of our limited understanding of their interaction and because of the complexity and heterogeneity of rangeland ecosystems. Carbon dynamics of croplands and their response to management systems or changes in cultural practices are quite simple in comparison to rangelands, because cultivated soil is more homogeneous than noncultivated systems, and its C inputs are larger and managed more uniformly.

This chapter discusses the effects of erosion and grazing on C dynamics and also tries to estimate the potential of this large land resource as a C sink.

Effects of Erosion

Soil erosion is one of the most visible drivers of C sequestration related to management in many systems. As Chapter 9 by Lal details, soil erosion can both increase and reduce carbon sequestration. Soil erosion exposes fixed organic carbon (OC) to higher oxidation rates by breaking the bonds which physically protect soil organic matter (SOM) (Gregorich et al., 1997). Organic matter (OM) which is not oxidized, however, may be deposited in another part of the landscape in which SOM turnover rates are lower than in the source location (Gregorich et al., 1998). This effectively increases C sequestration rates.

Much of the western U.S. drains into closed basins. Runoff from these rangelands collects in lower parts of the landscape in ephemeral lakes or playas. The soils in these areas often have higher SOM contents (see Table 5.1 in Ch. 5, Bird et al.). It is not clear, however, if this is due to the higher rates of net primary production (Huenneke, 1995), to differences in turnover rates due to soil texture, temperature, or moisture regime, or to redeposition of SOM from surrounding rangeland.

The moisture regime of the depositional area, in particular, has a significant impact on C storage. Rabenhorst (1995) showed that C density was eight times higher on a poorly drained soil than on a well drained soil formed from the same parent material. In hydrologically open systems, the SOM removed from rangelands may be deposited in marine sediments where the likelihood of reoxidation is extremely low (Fan et al., 1998).

The C movement associated with soil erosion in native rangelands tends to be proportionally higher than that in croplands due to the fact that OM is more highly concentrated in the top few millimeters of the rangeland soils and the steady-state level of SOM is higher in rangeland than cropland, given the same soil. The effects of erosion in high altitude and high latitude regions are magnified by the accumulation of an organic horizon at the soil surface, which frequently is underlain by permafrost.

Livestock management affects soil erosion in at least four ways (Fig. 11.1):

1. Hoof action generates surface disturbances which increase erodibility both directly and indirectly.
2. Hoof action incorporates surface litter and standing dead material into the soil, potentially increasing SOM and reducing soil erodibility.
3. Grazing reduces canopy cover directly.
4. Grazing can lead to changes in species composition.

Soil surface disturbance

Disturbance of physically and biologically crusted soils significantly reduces resistance to both wind and water erosion. Belnap and Gillette (1998) found that simulated disturbance by cattle significantly reduced threshold friction velocities on a variety of rangeland soils and that the effect of disturbance varied as a function of texture and of the development of the biological crust (Table 11.1).

A recently completed experiment conducted on a similar suite of soils showed that a disturbed soils' recovery of resistance to wind erosion is a function of rainfall intensity (J.E. Herrick, unpublished data). Consequently, the time of disturbance relative to aeolian and precipitation events may be as important as the type or intensity of the disturbance. Soil surface disturbance during grazing also increases susceptibility to water erosion (Weltz and Wood, 1986; McIvor et al., 1995).

Incorporation of aboveground material

The effect on production of incorporating litter and standing dead material into the soil, relative to the destruction of physical and biological crusts, has been debated hotly but has received very little scientific attention. Those who argue for a net benefit of animal impact also tend to view any increase in soil erosion as either minimal, benign, or both (Savory, 1988). Others (e.g., Belnap, 1995) argue that the costs of increases in erodibility far outweigh the benefits of incorporating OM, particularly in systems in which biological crusts largely stabilize the soil surface. Very few experiments have been designed to separate the effects of increased OM inputs to the soil surface from other changes associated with grazing.

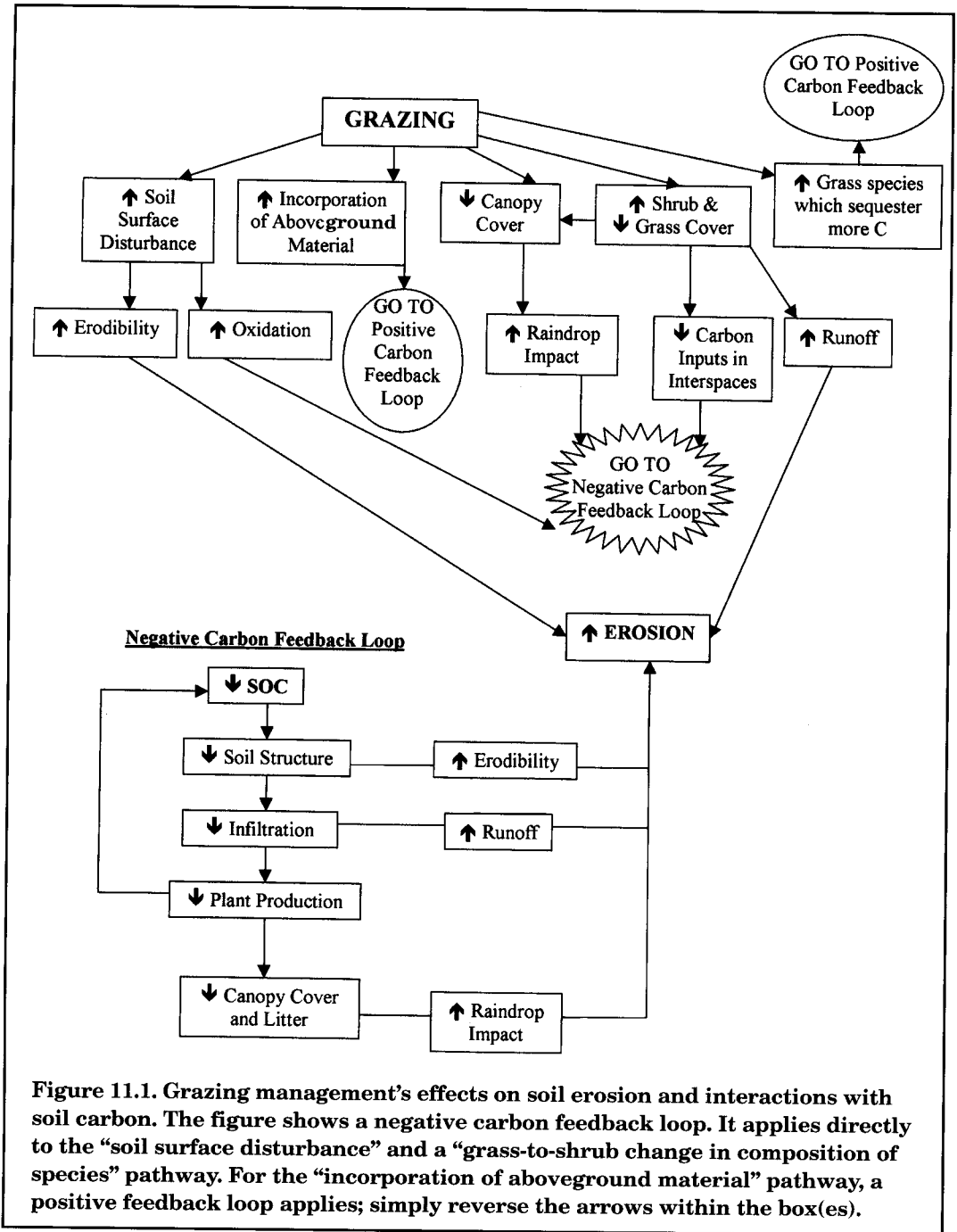


Figure 11.1. Grazing management's effects on soil erosion and interactions with soil carbon. The figure shows a negative carbon feedback loop. It applies directly to the "soil surface disturbance" and a "grass-to-shrub change in composition of species" pathway. For the "incorporation of aboveground material" pathway, a positive feedback loop applies; simply reverse the arrows within the box(es).

Table 11.1. Percent reduction of threshold friction velocities (TFV) following trampling by cattle for four sites in the northern Chihuahuan Desert (from Belnap and Gillette, 1998).

Site	Crust Type	% Sand	% Silt	% Clay	TFV Reduction
Sand	Well Developed Cyanobacterial	72.4	14.3	13.3	33-81
Gravel	Weakly Developed Cyanobacterial	68.7	11.6	19.6	22
Playa	Mineral	33.1	49.3	17.6	86
Silt	Well-Developed Cyanobacterial & Lichen	30.1	40.6	29.3	60

Canopy cover reduction

A number of studies have demonstrated that, in arid and semiarid rangelands in which vegetative cover rarely reaches 100%, total ground cover is the single most important factor affecting sediment production by water (Wood et al., 1987; Blackburn and Pierson, 1994). Reductions in canopy cover by grazing can approach 100%, depending on the management system (Savory, 1988). Other management activities also can increase the susceptibility of soil to wind erosion, through a combination of canopy cover reductions and increases in erodibility. Stout and Zobeck (1998) found that threshold wind speeds still were increasing 6 months after a fire burned a grass-dominated rangeland near Lubbock, Texas.

Changes in species composition

While the factors discussed above can have significant short-term effects on erosion and subsequent loss of C from rangelands, grazing's effects of changing the plant community structure and composition are more important over the long term. Using isotopes, researchers have correlated episodic erosional periods, recorded in soil profiles in the Chihuahuan Desert, with the replacement of C₄ grasses by C₃ shrubs (Monger et al., 1998).

This prehistoric record is reinforced by historic increases in soil erosion associated with a transition from a predominantly grass-dominated (*Bouteloua* spp., *Sporobolus* spp. and *Aristida* spp.) landscape to one dominated by mesquite (*Prosopis glandulosa*) and creosotebush (*Larrea tridentata*) (Buffington and Herbel, 1965). Follett et al. (1997), using stable C isotopes, also reported a shift from a predominately warm season (C₄) plant community to a more cool season (C₃) shrub plant community near Lawton, Oklahoma, and Big Springs, Texas.

The current transition results from an interaction between drought and the change in the disturbance regime associated with high levels of livestock grazing beginning at the end of the 19th century (Schlesinger et al., 1990). Although grazing by livestock can trigger these transitions, grazing management alone, or even the complete removal of livestock, is often insufficient to reverse the changes once they have begun. A total of 4.6 cm of soil was lost from one pasture over a 45-year period during which there was no grazing by domestic livestock (Gibbens et al., 1983). At the beginning of the study, the pasture was covered by a mix of black grama grass (*Bouteloua eriopoda*) and mesquite, with some areas already dominated by mesquite. By the end of the study, the complete area (over 250 ha) was covered by coppice sand dunes formed around mesquite plants.

These increases in erosion induced by vegetative changes are not necessarily associated with a reduction in C storage and, in fact, actually may increase C sequestration, at least over the short term. Over longer periods, however, exposure of the calcic horizon at the surface by erosion may lead to a net reduction in C storage, due to increased losses of carbonate (Ch. 4, Monger). This redistribution of SOM and other resources to create patches of different sizes has been discussed for other rangeland ecosystems (e.g., Tongway and Ludwig, 1990).

Net impact on C sequestration

The impact of soil erosion on C sequestration in source areas of sediment is clearly negative. In severely degraded areas in which both net primary productivity and plant canopy cover are reduced, C sequestration is obviously negative at the landscape scale. The net impact of erosion on C sequestration in less highly degraded rangeland systems is less clear and cannot be separated from feedbacks within the plant community (Fig. 11.1). The net impacts depend on the fate of the displaced C, on subsequent changes in the plant community, and on net primary productivity at both source and deposition areas.

In a global analysis, Lal (1995) assumed that up to 20% of displaced SOM mineralizes each year and is lost as CO₂. Few quantitative estimates of mineralization rates are available, and no landscape level studies on the net impacts of soil erosion have included burial and ecosystem level changes in net primary productivity. Net changes in C sequestration associated with plant community changes, however, have been measured. These figures integrate all sources of change in sources and sinks of C, including erosion.

Effects of Grazing

Evaluations of grazing's effects on soil C changes have varied in results. Milchunas and Lauenroth (1993) evaluated 236 data sets compiled worldwide that compared the effects of grazing on species composition, aboveground net primary production, root biomass, and soil nutrients. Of the many studies they reviewed, only 34 compared the effects of grazed vs. ungrazed (exclosure) areas on soil C. Of these 34, about 40% showed an increase in soil C in response to grazing and about 60% showed a decrease or no response to grazing.

These 34 studies represent research reported from 1947 to 1990 from throughout the world. Burke et al. (1997) further reviewed the biogeochemistry of rangelands in central North America and highlighted the belowground C's varied and inconsistent response to management.

Current research

Significant research has been completed since 1990, and it also shows considerable variance in C's response to grazing. Most of this research has compared systems with various grazing intensities and strategies to small ungrazed exclosures.

Bauer et al. (1987) compared relict rangelands and nearby grazed rangelands and reported that differences in soil C across soil texture and sampling depths were not consistent among grazing treatments. However, when averaged across soil textures and depths, the relict rangelands had 1.27 kg C/m² more in the 0.46 m soil depth than in the grazed areas, but soil N was higher in the grazed areas. They concluded that, since fencing of the rangelands in that part of the Northern Great Plains probably occurred about 75 years before their study, the differences would account for about 165 kg C/ha/yr of C loss due to grazing.

C vs. N response

The difference in C and N response to grazing is puzzling, since grazing has been shown to increase carbon and nitrogen cycling in rangeland systems. Grazing has been shown to increase the amount of medium litter, fine OM, and very fine OM in mixed grass prairie and parkland fescue sites (Naeth et al., 1991). Shariff et al. (1994) also reported that moderate grazing resulted in higher decomposition and soil N mineralization and lower N releases via decomposition than that observed on the long-term, nongrazed, and heavily grazed treatments. Litter and root decomposition averaged 55% on the moderately grazed treatment, 13% on the nongrazed, and 19% on the heavily grazed treatments. They suggested "that moderate grazing may lead to a greater conservation of N since this treatment had a higher level of N retention in OM (litter and dead roots)."

Enhanced N status has been shown to increase the rate of C and N mineralization and the total OC in the surface 2.5 and 7.5 cm of the soil in grasslands reestablished on marginal, highly erodible croplands (Reeder et al., 1998a). Frank et al. (1995) reported that moderate grazing (2.6 ha/steer) reduced the soil OC by 17% in the 107-cm soil profile, compared to that of the nongrazed enclosure they studied. However, they, like Dormaar and Willms (1990) and Smoliak et al. (1972), attributed the increase in soil C under heavy grazing to a change in plant community composition, which translated to a shallower and more robust root system of the C_4 species which replaced the more deeply rooting C_3 species.

Change in plant community

Grazing intensities that cause dramatic changes in a plant community's composition generally are identified as heavy or excessive and should be avoided to prevent reducing the quality of the resource, reducing production, reducing seasonality of the forage, and potentially reducing the sustainability of the system. Season-long grazing also can harm certain components of the plant community. Season-long grazing and/or heavy grazing reduce, or nearly eliminate, desirable C_3 species and shrubs from rangeland ecosystems (Schuman et al., 1999; Manley et al., 1997; Schuman et al., 1990; Smoliak et al., 1972).

Dormaar and Willms (1998) demonstrated that grazing at light stocking intensities (1.2 AUM/ha) did not have any effect on the SOC content after 44 years of grazing; however, heavy (2.4 AUM/ha) and very heavy (4.8 AUM/ha) grazing significantly reduced the SOC in the foothills of southwestern Alberta. They believe the heavy grazing intensities "jeopardized the sustainability of the ecosystem by reducing the fertility and water holding capacity."

Grazing intensity and strategy

As noted earlier, grazing does not always reduce SOC. Berg et al. (1997) evaluated the effects, in a native sandhill rangeland in western Oklahoma, of 50 years of grazing on vegetation and soil C and N. They evaluated pastures that were grazed at a moderate stocking intensity (0.29 yearling steer/ha/yr or 0.14 cow-calf pairs/ha/yr) on a year-long basis. This grazing intensity was defined as the level of grazing that left 1/3 of the average forage production at the end of the grazing period. They also evaluated three soil sampling procedures to determine their relative effects on the evaluation of soil C and N. They found that grazing significantly increased soil C mass in the surface 5 cm of the profile compared to that in the nongrazed enclosures but felt that a significant portion of this increase was due to the difference in bulk density (grazed 1.35 g/cm³ vs. ungrazed 1.19 g/cm³).

When sampling to a constant mass (5 cm deep or less) or sampling to a constant 5-cm depth and expressing the soil C on a concentration basis, the differences exhibited between grazed and ungrazed pastures were not significant at $P \leq$

0.05. In both cases, the concentration of SOC was still 10 and 17% greater in the grazed than in the nongrazed treatments, whether sampled to a constant mass or to a constant depth.

Berg et al. (1997) concluded that long-term grazing at the moderate stocking rate resulted in significant changes in plant community composition (based on frequency measurements) and that, as Hobbie (1992) suggested, "these changes are both a cause and effect of differences in nutrient cycling." Dormaar et al. (1984) also reported higher OC in the grazed than in the ungrazed treatments at Manyberries (mixed grass association) and Stavely (fescue grassland association), Alberta, with the exception of late winter/early spring at the Stavely location.

In southeast Wyoming at the USDA-ARS, High Plains Grasslands Research Station, SOC was significantly greater in the surface 30 cm of mixed grass rangeland pastures grazed season-long for 12 years at a light stocking rate (22 steer-days/ha) than in the nongrazed exclosures (Manley et al., 1995). SOC also was higher in pastures heavily grazed (67 steer days/ha) using grazing strategies of continuous season-long, rotationally deferred, and short duration, than in the nongrazed exclosures, when evaluating only the surface 15 cm of the soil (Manley et al., 1995).

In collateral studies, Schuman et al. (1999) evaluated the C balance of this mixed grass prairie site and concluded that the C mass in the plant-soil (30 cm depth) system was significantly greater in the continuous, season-long, light and heavy grazing treatments than in the ungrazed exclosures. Evaluation of the 0- to 60-cm soil-plant system showed that 89% to 93% of the system C was stored in SOM within the soil profile. Less than 10% of the C was found in the vegetative components (above- and belowground). The heavy stocking rate altered the plant community's composition, which could account for a portion of the change in the distribution of carbon among the system's components.

Schuman et al. (1999) concluded that the livestock's hoof action helped break down and incorporate the standing dead biomass and litter into the soil and enhanced biological decomposition, thereby reducing the potential loss of C from the system via photochemical oxidation. Blue grama, with a typically dense but shallow rooting system, showed an increase in root biomass under heavy grazing, while the western wheatgrass showed a decrease; however, no differences in root biomass were observed between the nongrazed, lightly grazed, and heavily grazed treatments. Therefore, root biomass differences could not be used to explain the increase in C.

Further studies were completed at this site to delineate the reasons for the greater C observed in grazed pastures than in ungrazed exclosures. Research evaluating the effects of livestock grazing on CO₂ exchange rates (CER) was completed using a closed chamber system (LeCain et al., 2000). The CER values were

adjusted to account for soil respiration. Higher early season CER was observed on the grazed pastures than on the ungrazed exclosures.

The CER on the grazed pastures was 1.5 to 2 times higher from April through June than on the exclosures. This higher early season CER related to earlier spring greening in the grazed pastures. Schuman et al. (2000) note that this could account for a portion of the increased soil C noted in the grazed treatments because, by the end of June, 60% to 80% of the aboveground production has occurred. April through June also is the period of optimum soil moisture.

Studies similar to those by Schuman et al. (1999) also were conducted at the USDA-ARS Central Plains Experimental Range in northeastern Colorado on the short grass steppe and found greater SOC under a heavily grazed pasture than in an adjacent exclosure after 50 years of grazing (Reeder et al., 1998b). However, a similar response to grazing was not evident on the lightly grazed treatment and its adjacent control. The lack of consistency in this case was attributed to the fact that, even though the soil series was carefully selected to be the same, the clay contents in the A and B horizon were significantly greater on the lightly grazed pasture and its exclosure.

Derner et al. (1997) also evaluated the effect of grazing on the short grass steppe in northeastern Colorado and found that SOC content was 43% and 55% higher in the 0- to 5- and 5- to 15-cm depth on the grazed than on the ungrazed treatment. However, they did not find this same relationship with grazing in the tall grass and midgrass communities they studied.

Grazing vs. mowing

Hassink and Neeteson (1991) compared the effects of grazing and mowing on the SOC status of the soil. After 4 years of treatment, they found that the amount of soil C was significantly greater in the top 5 cm of the soil under grazing than under mowing, for both a sandy and loamy soil. They concluded that, under grazing conditions, more organic material returns to the soil than under mowing, and dung and urine also return to the soil and result in more rapid cycling of C. They also concluded that, through the livestock's hoof action, grazing increases the amount of aboveground herbage incorporated into the soil.

Explanations for varied responses to grazing

The effects of grazing on SOC seem to be inconsistent and variable, and numerous explanations have been offered. Schuman et al. (1999) and Berg et al. (1997) identified factors that require careful consideration in this discussion.

Methodology, soil, and climate

Schuman et al. (1999) pointed out that consistent sampling procedures, including the handling of surface litter, comparable soil series/characteristics, similar baseline plant communities, and consistent laboratory methodologies are essential to ensuring sound treatment evaluations. Berg et al. (1997) expanded the list to include reporting on a concentration basis and ensuring similar and appropriate sampling depths and valid field replication.

Considerable discussion has occurred over reporting concentration, rather than correcting for differences in bulk density, to report quantity (Skene, 1966; Henzell et al., 1967). Berg et al. recommended comparing adjacent pastures, to show contrasts in plant communities induced by grazing, as a good approach to determining grazing's effects on soil C.

Rauzi et al. (1968) used a similar approach to characterize infiltration rates in the Great Plains. Dormaar et al. (1977) also found that total C and N, C:N ratios, and numerous other SOM characteristics depend greatly on the season of sampling. Precipitation variation among years also has as great or greater effects on a plant community's response and annual net primary productivity as do grazing treatments (Michunas and Lauenroth, 1993).

Plant response to grazing

Many physiological characteristics of plants also may help explain the differences various researchers observed. Caldwell et al. (1981) and Dormaar et al. (1995) point out that species differ greatly in their response to grazing and, hence, soil C's responses may differ greatly. Crested wheatgrass (*Agropyron desertorum*) has shown greater flexibility of resource allocation for C following defoliation, with more going to the root system with curtailed root growth, whereas root growth in *A. spicatum* continued at the same rate after as before defoliation (Caldwell et al., 1981).

Photosynthesis of grazed plants

Much direct and indirect evidence has shown that grazing or simulated defoliation influences the rate of photosynthesis and that the actual interaction between the grazer and the plant is important (Painter and Detling, 1981; Dyer and Bokhari, 1976; Detling et al., 1979). Painter and Detling (1981) simulated a mod-

erate and heavy defoliation of western wheatgrass plants and found that net photosynthesis was greater on those plants than on the control plants.

Others have reported the enhancement of photosynthesis in the undamaged leaves of partially defoliated plants, which is thought to result from decreased mesophyll's resistance to CO₂'s diffusion in new leaves. A single, simulated grazing of blue grama resulted in greater net photosynthesis, and it also appears that the increased photosynthesis results from increased photosynthetic capacity of the tissues that remained or were produced after defoliation (Detling et al., 1979).

Plant growth

Detling et al. (1979) reported that total C losses from root and crown respiration were less in the clipped plants because of their reduced biomass. Hodgkinson and Baas Becking (1977) reported that, when shoots are grazed, root mortality frequently increases and root extension and branching decreases. This root mortality could result in significant increases in SOC over a relatively short time. Stevenson and White (1941) reported greater root production under grazed than ungrazed prairie, even though many grazing studies simulated in greenhouses have shown decreased root growth (Johnston, 1961; Wilson, 1960).

The physical and biochemical effects of grazing by livestock and insects have greater effects on regrowth than simulated grazing. Dyer and Bokhari (1976) reported that letting grasshoppers feed on hydroponically grown blue grama resulted in much greater regrowth potential than simply clipping plants, partially because of the increase in tiller production.

Schuman et al. (1990) also showed increased rhizome production of western wheatgrass under grazing. Dyer and Bokhari (1976) reported that grasshoppers' grazing resulted in an increase in belowground respiration and root exudation. Grasshoppers' grazing increased the rate of change in pH of the root medium by 4% per day, and they believe this is the result of increased respiration of the roots (release of CO₂) and the production of organic acids in the root exudates. They state that this increase "could well affect the total metabolism and total biomass of the plants to a far greater extent than direct consumption or litter production." The observed effects on tiller production could have a significant impact on later growth potentials.

Jameson (1964) and Reardon et al. (1972, 1974) believe that cattle's grazing on grass also stimulates metabolic activities through the mechanical action and absorption of saliva into the grass plant. Reardon et al. (1974) believe that the thiamine in saliva may be important because of coevolutionary considerations between the herbivore and the producer. Dormaar (1988) reported that the growth of blue grama and rye (*Secale cereale*) significantly affected the soil pH, and that blue grama had a slightly greater effect than the rye.

Complexity of issues

The variable responses of soil C to grazing is complex and cannot be attributed solely to methodological variations and constraints. Variations in soil C in response to management alternatives undoubtedly reflect the complexity of the soil C balance and the many, poorly understood, indirect effects of grazing (Table 11.2).

Table 11.2 summarizes the literature this chapter cites that evaluates the effects of grazing on soil C. As noted earlier, erosion induced by grazing, even in very limited magnitude, could have major impacts on soil C because of the surface soil material's importance in C storage and activity and its susceptibility to erosion.

Table 11.2. Responses of grassland soil C to grazing.		
Carbon Response	Reference	Comment(s)
Increase in C	Milchunas & Lauenroth, 1993	14 sites, grazed
	Frank et al., 1995	Heavy grazing, PCC
	Dormaar & Willms, 1990	Heavy grazing, PCC
	Schuman et al., 1999	Light & heavy grazing
	Manley et al., 1995	Light & heavy grazing
	Berg et al., 1997	Moderate grazing
	Dormaar et al., 1984	Grazed, PCC
	Reeder et al., 1998b	Heavy grazing, PCC
	Derner et al., 1997	Grazed, short grass
	Smoliak et al., 1972	Heavy grazing, PCC
Decrease/No Change in C	Milchunas & Lauenroth, 1993	20 sites, grazed
	Bauer et al., 1987	Moderate grazing, increased N
	Frank et al., 1995	Moderate grazing, increased N
	Dormaar & Willms, 1998	Light grazing (no change) heavy & very heavy grazing (reduced soil C)
	Reeder et al., 1998b	Light grazing (no change)
	Derner et al., 1997	Grazed, no change tall- & mid-grass
<i>PCC = plant community change with increased root biomass resulting.</i>		

Rangelands' Potential to Sequester Carbon

Most rangeland soils are probably at or near an equilibrium state, if the vegetative community and conditions have been consistent for many years. They are an important C storage zone, but their potential for enhanced C storage may be limited under these equilibrium conditions.

Many of the proposed effects of management on C storage in rangelands assume constant environmental conditions. In fact, we know that climate and environment are not static, and they may be changing at an accelerated rate because of human activity. One of the most profound changes is the increase in atmospheric CO₂. Current concentrations are already about 30% higher than in pre-industrial times, and concentrations are increasing at a rate of 1.5 ppmv per year (Houghton, 1997).

Increases in atmospheric CO₂ can affect rangeland C storage in several ways. First, elevated CO₂ can accelerate rates of photosynthesis directly, thereby increasing biomass production (Owensby et al., 1994; van Ginkel and Gorrisson, 1998). Increases in assimilated C may be allocated preferentially below ground (Morgan et al., 1994). Increases in atmospheric CO₂ also may enhance yields by improving the efficiency of water use (Owensby, 1993; Morgan et al., 1994).

Elevated atmospheric CO₂ also may slow the rate of decomposition, which in turn would increase C in the system (Owensby, 1993). Research has shown that roots of grasses grown under enriched CO₂ decompose more slowly than those of plants grown under ambient CO₂ levels (Gorrison et al., 1995; van Ginkel et al., 1996; van Ginkel and Gorrison, 1998). Consequently, increases in atmospheric CO₂ may enhance SOC both by increasing the amount of plant litter returned to the soil through increases in net primary production, and by slowing its decomposition within the soil. Owensby (1993) suggests that doubling the atmospheric CO₂ over 50 to 70 years might elicit a 12% to 15% gain in SOM under tall grass prairie.

The effect of elevated CO₂ cannot be evaluated independently of other concomitant changes. Increases in temperature may accelerate microbial decomposition of OM (Jenkinson et al., 1991; Raich and Schlesinger, 1992; Trumbore et al., 1996), thereby offsetting some of the possible gains in stored C from increased atmospheric CO₂. A further confounding factor is possible accelerated N mineralization with higher temperatures, which could enhance productivity and ameliorate the loss of C (Schimel, 1995). Therefore, the net impact of a changing atmosphere will reflect the interaction of numerous factors (Lashof et al., 1997).

Enhanced deposition of humanly induced N changes will have a significant effect on C storage in rangelands. The amount of N₂ fixed by human activities already rivals or exceeds that from natural sources (Vitousek et al., 1997), and further increases are expected (Jeffries and Maron, 1997). On rangelands, most of which are limited in N, deposition of supplemental N can promote productivity

and enhance C storage (Wedlin and Tilman, 1996). The magnitude of gains in terms of C storage will vary with the plant community, which the degree of N deposited also will affect.

N deposition has been suggested as an important mechanism of terrestrial C gain (Schindler and Bayley, 1993). Some short-term fertilization studies have suggested possible effects of N deposition on C dynamics (Reeder et al., 1998a); however, the long-term impact of low-level increases in N on a wide range of rangeland communities remains uncertain.

Estimate of Potential C Gains

It should be obvious from the above discussion that estimating potential C gains is more complicated for rangeland soils than for cultivated lands. Though grazing lands occupy about twice the area of croplands in the U.S., comparatively few long-term studies have been made of soil C's dynamics in grazing land. Impacts of grazing land management on soil C tend to be less abrupt than those on cultivated land. Grazing land soils have inherently high C densities, and these C stocks change relatively slowly because management's effects, like changes in grazing strategy, often are subtle and imposed gradually.

Finally, rangeland ecosystems are much more complex than most croplands. Whereas plant communities of single species are imposed on croplands, rangeland communities are diverse and evolved over time in response to climate, soils, and management practices. Consequently, a shift in management may induce a gradual change in species composition, and the eventual impact of that change and its various feedbacks on soil C may not be measured until years or decades later.

Because of these complexities, the potential rates of C sequestration on rangelands can only be qualitative estimates. Nevertheless, given the vast area and large C pool involved, it is important to at least derive some preliminary estimates of possible changes in C storage.

Assumptions

To estimate the potential gain of C in U.S. rangelands, we make the following assumptions:

1. Under consistent management and environment, soil C reserves in any ecosystem eventually approach a steady-state value, beyond which, rates of soil C gain or loss are negligible from the standpoint of the atmospheric CO₂ pool (Odum, 1969; Johnson, 1995).

2. Change in the current trend of net C exchange depends on a shift in management or environmental conditions. Thus, the C content of the soil currently at a steady state can be altered appreciably only by a shift in management or climate. Similarly, a soil currently losing C, approaching a new, lower steady state, can be shifted from that course only by a change in management or other external factor(s).
3. From the perspective of reducing atmospheric CO₂ gains, an avoided loss of soil C is as important as a gain in soil C (Izaurre et al., 1999). A management choice that avoids losing 1 MMT of soil C has exactly the same benefit to atmospheric CO₂ as a management choice that gains 1 MMT of soil C.

Estimated benefits

These assumptions make it possible to derive at least conceptual estimates of the benefits of good rangeland management for mitigating increases in atmospheric CO₂ (Table 11.3).

According to recent estimates, about one-third of the U.S. rangeland area (54 Mha) has no serious ecological or management problem (USDA, NRCS, 1998; USDI, BLM, 1998; David Wheeler, USDA, FS, Lakewood, CO, personal communication, 1999). If we assume that the soil C content of these rangelands is at steady state, then these soils have little potential for further C storage gains (assumption 1). But the remaining two-thirds of the U.S. rangelands, presumably, have some constraints which limit productivity and, hence, C storage.

The potential C gain in these lands varies widely, but any improvements in soil C may be gradual, perhaps in the area of 0.1 MTC/ha/yr. Much faster rates of C storage gain are possible in previously cultivated lands reseeded to grass. Bruce et al. (1999) estimated that "set aside" lands under the Conservation Reserve Program (CRP) are gaining C at a rate of about 0.6 MTC/ha/yr, yielding a total rate of C sequestration of about 8 MMTC/ha/yr. Thus, improved management may sequester up to 11 MMTC/ha/yr in permanent rangelands and another 8 MMTC/ha/yr in recently reestablished grasslands (Table 11.3).

Equally as important as the potential C gains are any averted losses (assumption 3) (Table 11.3). Because rangelands represent such a large pool of C, preserving existing reserves is especially important. One way to avoid losses is to maintain or establish optimal grazing strategies. Recent studies have shown that properly grazed rangeland can gain C at a rate of about 0.3 MTC/ha/yr, relative to that in a corresponding ungrazed enclosure (Schuman et al., 1999; Manley et al., 1995). Consequently, maintaining good grazing strategies on all well managed rangelands, rather than ceasing grazing, as some advocate, would yield an avoided loss of about 16 MMTC/ha/yr.

Table 11.3. Estimated potential benefits to the mitigation of atmospheric CO₂ from the adoption of improved management of grasslands and potential avoided losses from grasslands in the U.S.

Land Use	Area ⁴ (Mha)	Rate ⁵ (MTC/ha/yr)	Rate ⁵ (MMTC/ha/yr)
Potential Mitigation Gains			
Well managed grasslands ¹	54	0	0
Poorly managed grasslands ¹	107	0.1	11
CRP grasslands ²	13	0.6	8
		Total Gain	19
Potential Avoided Losses			
Well managed grasslands ¹	54	0.3	16
Poorly managed grasslands ¹	107	0.2	21
CRP grasslands ³	13	0.3	4
		Total Avoided Loss	41
<p>¹ According to USDA-NRCS (1998), 33% (54 Mha) of the private U.S. rangelands are reported to have no serious ecological or management problems, and 67% (109 Mha) would benefit from enhanced management or restoration. Of federally managed rangelands under the authority of the Bureau of Land Management, 37% (2 Mha) are in good to excellent condition and 63% (4 Mha) are in fair to poorer condition (USDI-BLM 1998). 80% (0.8 Mha) of the National Grasslands in the Rocky Mountain Region managed by the USDA-FS meet the forest plan objectives, and 20% (0.2 Mha) are considered moving toward those objectives, not meeting or moving toward them, or undetermined (David Wheeler, Rocky Mountain Regional USDA-FS, Lakewood, CO, personal communication, 1999).</p> <p>² CRP = Conservation Reserve Program; area and rate of potential C gain from Bruce et al. (1999).</p> <p>³ Rate of avoided loss (0.3 MTC/ha/yr) based on rate measured by Doran et al. (1998) at site converted from established grassland to a no-till wheat-fallow system.</p> <p>⁴ Land area values are those compiled by Sobecki et al., Ch. 2, and are used to provide consistency throughout this volume. However, the values in footnote #1 above are those the respective agencies published in their inventory reports.</p> <p>⁵ Potential rates apply to possible changes in the period soon after a management change (or in the case of CRP lands, to current rates on 'existing' lands). Rates of accrual will diminish with time (perhaps after a few years or decades) as soil C approaches a new steady state.</p>			

This value assumes the net gain relative to losses incurred if grazing were discontinued on all well managed rangelands, so it represents a maximum value. Similarly, avoided losses by maintaining current grazing practices on "poorly managed" rangelands might amount to as much as 23 MMTC/yr.

Finally, preventing cultivation of existing CRP lands can prevent potentially large losses of stored C. Assuming that conversion of CRP lands to cropland would result in C loss at a rate of 0.3 MTC/ha/yr (Doran et al., 1998), preserving all CRP lands in rangelands avoids a potential loss of 4 MMTC/ha/yr. This avoided loss is in addition to the rates of C accrual at these sites, assumed to be 0.6 MTC/ha/yr. Potential avoided loss from all these sources therefore totals about 43 MMTC/ha/yr. This value does not yet include any losses avoided by preventing further cultivation of existing, reestablished, or native rangelands.

These estimates are just that, estimates, and perhaps better viewed as conceptual illustrations to show the magnitude of potential C sequestration or avoid-

ed losses. Nevertheless, they demonstrate that the vast C pool held in U.S. rangelands cannot be ignored in assessing the link between agricultural management and atmospheric CO₂. If nothing else, they illustrate the importance to atmospheric CO₂ of preventing disturbance of these vast C-rich systems.

Conclusion

Carbon dynamics of rangeland soils are very complex, and only limited detailed research has been conducted to assess the role of management and grazing strategies on soil C changes and the potential for enhanced C sequestration. Soil erosion also can affect the soil C pools in rangeland ecosystems, but much of the erosion does not result in loss of C from the landscape but rather a repositioning on the landscape. However, a significant portion of this redeposited SOM may undergo mineralization and subsequently be lost from the system.

The U.S. rangelands are already a large repository of C because of their high C density and vast land area. Improved practices on poorly managed rangelands could further increase this C store; our estimate suggests a theoretical accumulation rate of 11 MMTC/ha/yr, though this might be a theoretical maximum. A further gain of 8 MMTC/ha/yr might accrue from maintaining existing CRP lands. But perhaps the most important management opportunity, from the standpoint of atmospheric CO₂, is the preservation of current stores of C in U.S. rangelands. For example, conservation practices might prevent losses amounting to as much as 43 MMTC/ha/yr.

All of these estimates perhaps are viewed best as conceptual illustrations rather than quantitative values. Nevertheless, they illustrate the significance of U.S. rangelands as a C storehouse, and the importance of management to preserve or even enhance their rich C reserves. As more directed research specifically addresses the effects of management on C storage, and as CO₂ flux measurements become available, a more thorough and accurate estimation of C sequestration by rangelands will be possible.

References

- Batjes, N.H. 1996. Total carbon and nitrogen in the soils of the world. *Eur. J. of Soil Sci.* 47:151-163.
- Bauer, A., C.V. Cole, and A.L. Black. 1987. Soil property comparisons in virgin grasslands between grazed and nongrazed management systems. *Soil Sci. Soc. of Am. J.* 51:176-182.

- Belnap, J. 1995. Surface disturbances: their role in accelerating desertification. *Environ. Monit. and Assess.* 37:39-57.
- Belnap, J., and D.A. Gillette. 1998. Vulnerability of desert biological crusts to wind erosion: the influences of crust development, soil texture and disturbance. *J. of Arid Environ.* 39:133-142.
- Blackburn, W.H., and F.B. Pierson, Jr. 1994. Sources of variation in interrill erosion on rangelands. In Wilbert H. Blackburn, Frederick B. Pierson, Jr., Gerald E. Schuman, and R. Zartman (eds), *Variability in Rangeland Water Erosion Processes*, Soil Science Society of America, Madison, WI, pp. 1-10.
- Berg, W.A., J.A. Bradford, and P.L. Sims. 1997. Long-term soil nitrogen and vegetation change on sandhill rangeland. *J. of Range Manage.* 50:482-488.
- Bronson, K.F., K.G. Cassman, R. Wassmann, D.C. Olk, M. van Noordwijk, and D.P. Garrity. 1997. Soil carbon dynamics in different cropping systems in principal ecoregions of Asia. In R. Lal, J.M. Kimble, R.F. Follett, and B.A. Stewart (eds), *Management of Carbon Sequestration in Soil*, CRC Press, Boca Raton, FL, pp. 35-57.
- Bruce, J.P., M. Frome, H. Haites, H. Janzen, R. Lal, and K. Paustian. 1999. Carbon sequestration in soils. *J. Soil and Water Cons.* 54:382-389.
- Buffington, L.C., and C.H. Herbel. 1965. Vegetational changes on a semidesert grassland range from 1858 to 1963. *Ecol. Monogr.* 35:139-164.
- Burke, I.C., W.K. Lauenroth, and D.G. Milchunas. 1997. Biogeochemistry of managed grasslands in central North America. In E.A. Paul, K. Paustian, E.T. Elliott, and C.V. Cole (eds), *Soil Organic Matter in Temperate Agroecosystems, Long-term Experiments in North America*, CRC Press, Boca Raton, FL, pp. 85-102.
- Buyanovsky, G.A., and G.H. Wagner. 1998. Changing role of cultivated land in the global carbon cycle. *Biol. and Fertil. of Soils* 27:242-245.
- Caldwell, M.M., J.H. Richards, D.A. Johnson, R.S. Nowak, and R.S. Dzurec. 1981. Coping with herbivory: Photosynthesis capacity and resource allocation in two semiarid *Agropyron* bunchgrasses. *Oecologia* 50:14-24.
- Chadwick, O.A., E.F. Kelly, D.M. Merritts, and R.G. Amundson. 1994. Carbon dioxide consumption during soil development. *Biogeochemistry* 24:115-127.
- Derner, J.D., D.D. Briske, and T.W. Boutton. 1997. Does grazing mediate soil carbon and nitrogen accumulation beneath C₄ perennial grasses along an environmental gradient? *Plant and Soil* 191:147-156.
- Detling, J.K., M.I. Dyer, and D.T. Winn. 1979. Net photosynthesis, root respiration, and regrowth of *Bouteloua gracilis* following simulated grazing. *Oecologia* 41:127-134.
- Doran, J.W., E.T. Elliott, and K. Paustian. 1998. Soil microbial activity, nitrogen cycling, and long-term changes in organic carbon pools as related to fallow tillage management. *Soil & Tillage Res.* 49:3-18.
- Dormaar, J.F., A. Johnston, and S. Smoliak. 1977. Seasonal variation in chemical characteristics of soil organic matter of grazed and ungrazed mixed prairie and fescue grassland. *J. of Range Manage.* 30:195-198.

- Dormaar, J.F., A. Johnston, and S. Smoliak. 1984. Seasonal changes in carbon content, and dehydrogenase, phosphatase, and urease activities in mixed prairie and fescue grassland Ah horizons. *J. of Range Manage.* 37:31-35.
- Dormaar, J.F., M.A. Naeth, W.D. Willms, and D.S. Chanasyk. 1995. Effect of native prairie, crested wheatgrass (*Agropyron cristatum* (L.) Gaertn.) and Russian wildrye (*Elymus junceus* Fisch.) on soil chemical properties. *J. of Range Manage.* 48:258-263.
- Dormaar, J.F., and W.D. Willms. 1990. Effect of grazing and cultivation on some chemical properties of soils in the mixed prairie. *J. of Range Manage.* 43:456-460.
- Dormaar, J.F., and W.D. Willms. 1998. Effect of forty-four years of grazing on fescue grassland soils. *J. of Range Manage.* 51:122-126.
- Dyer, M.I., and U.G. Bokhari. 1976. Plant-animal interactions: Studies of the effects of grasshopper grazing on blue grama grass. *Ecology* 57:762-772.
- Eswaran, H., E. Van den Berg, P. Reich, and J. Kimble. 1995. Global soil carbon resources. In R. Lal, J. Kimble, E. Levine, and B.A. Stewart (eds), *Soils and Global Change*, CRC Lewis Publishers, Boca Raton, FL, pp. 27-43.
- Fan, S., M. Gloor, J. Mahlman, S. Pacaia, J. Sarmiento, T. Takahashi, and P. Tans. 1998. A large terrestrial carbon sink in North America implied by atmospheric and oceanic carbon dioxide data and models. *Science* 282:442-446.
- Follett, R.F., E.A. Paul, S.W. Leavitt, A.D. Halvorson, D. Lyon, and G.A. Peterson. 1997. Carbon isotope ratios of Great Plains soils and in wheat-fallow systems. *Soil Sci. Soc. Am. J.* 61:1068-1077.
- Frank, A.B., D.L. Tanaka, L. Hofmann, and R.F. Follett. 1995. Soil carbon and nitrogen of Northern Great Plains grasslands as influenced by long-term grazing. *J. of Range Manage.* 48:470-474.
- Gibbens, R.P., J.M. Tromble, J.T. Hennessy, and M. Cardenas. 1983. Soil movement in mesquite dunelands and former grasslands of southern New Mexico from 1933 to 1980. *J. of Range Manage.* 36:145-148.
- Gorrison, A., J.H. van Ginkel, J.J.B. Keurentjes, and J.A. van Veen. 1995. Grass root decomposition is retarded when grass has been grown under elevated CO₂. *Soil Biol. and Biochem.* 27:117-120.
- Gregorich, E.G., C.F. Drury, B.H. Ellert, and B.C. Liang. 1997. Fertilization effects on physically protected light fraction organic matter. *Soil Sci. Soc. of Am. J.* 61:482-484.
- Gregorich, E.G., K.J. Greer, D.W. Anderson, and B.C. Liang. 1998 Carbon distribution and losses: erosion and deposition effects. *Soil & Tillage Res.* 47:291-302.
- Hassink, J., and J.J. Neeteson. 1991. Effect of grassland management on the amounts of soil organic N and C. *Netherlands J. of Agric. Sci.* 39:225-236.
- Henzell, E.F., I.F. Fergas, and A.E. Martin. 1967. Accretion studies of soil organic matter. *J. Aust. Inst. of Agric. Sci.* 33:35-37.
- Hobbie, S.A. 1992. Effects of plant species on nutrient cycling. *TREE* 7:336-339.
- Hodgkinson, K.C., and H.G. Baas Becking. 1977. Effect of defoliation on root growth of some arid zone perennial plants. *Aust. J. Agric. Res.* 29:31-42.

- Houghton, R.A. 1995. Changes in the storage of terrestrial carbon since 1850. In R. Lal, J. Kimble, E. Levine, and B.A. Stewart (eds), *Soils and Global Change*, CRC Lewis Publishers, Boca Raton, FL, pp. 45-65.
- Houghton, J. 1997. *Global Warming: The Complete Briefing (2nd edition)*. Cambridge University Press, Cambridge. 251 pp.
- Huenneke, L.F. 1995. Shrublands and grasslands of the Jornada Long-Term Ecological Research Site: desertification and plant community structure in the northern Chihuahuan Desert. In J.R. Barrow, E.D. McArthur, R.E. Sosebee, and R.J. Tausch (compilers), *Proceedings: Shrubland Ecosystem Dynamics in a Changing Environment*, U.S. Forest Service Intermountain Research Station, Ogden, UT, pp. 48-50.
- Izaurrealde, R.C., K.H. Haugen-Kozyra, D.C. Jans, W.B. McGill, R.F. Grant, and J.C. Hiley. 2000. Soil organic carbon dynamics: measurement, simulation and site to region scale-up. In R. Lal, R.F. Follett, and B.A. Stewart (eds), *Advances in Soil Science: Assessment Methods for Soil C Pools*, CRC Press/Lewis Publishers, Boca Raton, FL (in press).
- Jameson, D.A. 1964. *Forage Plant Physiology and Soil-Range Relationships. Effect of Defoliation on Forage Plant Physiology. Special Publication 5*. American Society of Agronomy, Madison, WI. pp. 67-80.
- Jeffries, R.L., and J.L. Maron. 1997. The embarrassment of riches: atmospheric deposition of nitrogen and community and ecosystem processes. *TREE* 12:74-78.
- Jenkinson, D.S., D.E. Adams, and A. Wild. 1991. Model estimates of CO₂ emissions from soil in response to global warming. *Nature* 351:304-306.
- Johnson, M.G. 1995. The role of soil management in sequestering soil carbon. In Lal et al. (eds), *Soil Management and Greenhouse Effect*, Lewis Publishers, Boca Raton, FL, pp. 351-363.
- Johnston, A. 1961. Comparison of lightly grazed and ungrazed range in the fescue grassland of southwestern Alberta. *Can. J. Plant Sci.* 41:615-622.
- Lal, R. 1995. Global soil erosion by water and carbon dynamics. In R. Lal, J. Kimble, E. Levine, and B.A. Stewart (eds), *Soils and Global Change*, CRC Press, Boca Raton, FL, pp. 131-142.
- Lashof, D.A., B.J. DeAngelo, S.R. Saleska, and J. Harte. 1997. Terrestrial ecosystem feedbacks to global climate change. *Annu. Rev. Energy and Environ.* 22:75-118.
- LeCain, D.R., J.A. Morgan, G.E. Schuman, J.D. Reeder, and R.H. Hart. 2000. Carbon exchange of grazed and ungrazed pastures of a mixed grass prairie. *J. of Range Manage.* 53:199-206.
- Manley, W.A., R.H. Hart, M.J. Samuel, M.A. Smith, J.W. Waggoner, and J.T. Manley. 1997. Vegetation, cattle, and economic responses to grazing strategies and pressures. *J. of Range Manage.* 50:638-646.
- Manley, J.T., G.E. Schuman, J.D. Reeder, and R.H. Hart. 1995. Rangeland soil carbon and nitrogen responses to grazing. *J. of Soil and Water Cons.* 50:294-298.
- McIvor, J.G., J. Williams, and C.J. Gardener. 1995. Pasture management influences runoff and soil movement in the semiarid tropics. *Aust. J. of Exp. Agric.* 35:55-65.
- Milchunas, D.G., and W.K. Lauenroth. 1993. Quantitative effects of grazing on vegetation and soils over a global range of environments. *Ecol. Monogr.* 63(4):327-366.

- Monger, H.C., D.R. Cole, J.W. Gish, and T.H. Giordano. 1998. Stable carbon and oxygen isotopes in Quaternary soil carbonates as indicators of ecogeomorphic changes in the northern Chihuahuan Desert, U.S.A. *Geoderma* 82:137-172.
- Morgan, J.A., W.G. Knight, L.M. Dudley, and H.W. Hunt. 1994. Enhanced root system C-sink activity, water relations and aspects of nutrient acquisition in mycotrophic *Bouteloua gracilis* subjected to CO₂ enrichment. *Plant and Soil* 165:139-146.
- Naeth, M.A., A.W. Bailey, D.J. Pluth, D.S. Chanasyk, and R.T. Hardin. 1991. Grazing impacts on litter and soil organic matter in mixed prairie and fescue grassland ecosystems of Alberta. *J. of Range Manage.* 44:7-12.
- Odum, E.P. 1969. The strategy of ecosystem development. *Sci.* 164:262-270.
- Owensby, C.E. 1993. Potential impacts of elevated CO₂ and above- and below-ground litter quality of a tallgrass prairie. *Water Air Soil Pollut.* 70:413-424.
- Owensby, C.E., L.M. Auen, and P.I. Coyne. 1994. Biomass production in a nitrogen-fertilized, tallgrass prairie ecosystem exposed to ambient and elevated levels of CO₂. *Plant and Soil* 165:105-113.
- Painter, E.L., and J.K. Detling. 1981. Effects of defoliation on net photosynthesis and regrowth of western wheatgrass. *J. of Range Manage.* 34:68-71.
- Parton, W.J., J.W.B. Stewart, and C.V. Cole. 1988. Dynamics of C, N, P, and S in grassland soils: a model. *Biogeochem.* 5:109-131.
- Paustian, K., O. Andren, H.H. Janzen, R. Lal, P. Smith, G. Tian, H. Tiessen, M. Van Noordwijk, and P.L. Woomer. 1997. Agricultural soils as a sink to mitigate CO₂ emissions. *Soil Use and Manage.* 13:230-244.
- Rabenhorst, M.C. 1995. Carbon storage in tidal marsh soils. In R. Lal, J. Kimble, E. Levine, and B.A. Stewart (eds). *Soils and Global Change*, CRC-Lewis Press, Boca Raton, FL, pp. 93-103.
- Raich, J.W., and W.H. Schlesinger. 1992. The global carbon dioxide flux in soil respiration and its relationship to vegetation and climate. *Tellus* 44B:81-99.
- Rauzi, F., C.L. Fly, and E.J. Dyksterhuis. 1968. *Water intake of midcontinental rangelands as influenced by soil and plant cover. USDA Tech. Bull. 1390.* USDA. U.S. Government Printing Office. Washington, DC.
- Reardon, P.Q., C.L. Leinweber, and L.B. Merrill. 1972. The effect of bovine saliva on grasses. *J. of Anim. Sci.* 34:897-898.
- Reardon, P.Q., C.L. Leinweber, and L.B. Merrill. 1974. Response of sideoats grama to animal saliva and thiamine. *J. of Range Manage.* 27:400-401.
- Reeder, J.D., G.E. Schuman, and R.A. Bowman. 1998a. Soil C and N changes on conservation reserve program lands in the Central Great Plains. *Soil and Tillage Res.* 47:339-349.
- Reeder, J.D., G.E. Schuman, J.A. Morgan, D.R. LeCain, and R.H. Hart. 1998b. Impact of livestock grazing on the carbon and nitrogen balance of a shortgrass steppe. *Agronomy Abstracts.* American Society of Agronomy, Madison, WI. p. 291.
- Savory, A. 1988. *Holistic Resource Management.* Island Press. Washington, D.C.
- Schimel, D.S. 1995. Terrestrial ecosystems and the carbon cycle. *Global Change Biol.* 1:77-91.

- Schindler, D.W., and S.E. Bayley. 1993. The biosphere as an increasing sink for atmospheric carbon: estimates from increased nitrogen deposition. *Global Biogeochem. Cycles* 7:717-733.
- Schlesinger, W.H., J.F. Reynolds, G.L. Cunningham, L.F. Huenneke, W.M. Jarrell, R.A. Virginia, and W.G. Whitford. 1990. Biological feedbacks in global desertification. *Science* 247:1043-1048.
- Schlesinger, W.H. 1995. An overview of the carbon cycle. In R. Lal, J. Kimble, E. Levine, and B.A. Stewart (eds), *Soil and Global Change*, CRC-Lewis Press, Boca Raton, FL, pp. 9-25.
- Schuman, G.E., D.T. Booth, and J.W. Waggoner. 1990. Grazing reclaimed mined land seeded to native grasses in Wyoming. *J. of Soil and Water Cons.* 44:653-657.
- Schuman, G.E., D.R. LeCain, J.D. Reeder, and J.A. Morgan. 2000. *Carbon dynamics and sequestration of a mixed-grass prairie as influenced by grazing*. *Soil Sci. Soc. America, Special Publication*. Soil Sci. Soc. of Am. Madison, WI. (In press.)
- Schuman, G.E., J.D. Reeder, J.T. Manley, R.H. Hart, and W.A. Manley. 1999. Impact of grazing management on the carbon and nitrogen balance of a mixed-grass rangeland. *Ecol. Appl.* 9(1):65-71.
- Shariff, A.R., M.E. Biondini, and C.E. Grygiel. 1994. Grazing intensity effects on litter decomposition and soil nitrogen mineralization. *J. of Range Manage.* 47:444-449.
- Skene, J.K.M. 1966. Errors in accretion studies of soil organic matter. *J. Austr. Instit. of Agric. Sci.* 32:208-209.
- Smoliak, S., J.F. Dormaar, and A. Johnston. 1972. Long-term grazing effects on *Stipa-Bouteloua* prairie soils. *J. of Range Manage.* 25:246-250.
- Stevenson, T.M., and W.J. White. 1967. Root fibre production of some perennial grasses. *Sci. Agric.* 22:108-118.
- Stout, J.E., and T.M. Zobeck. 1998. Earth, wind and fire: aeolian activity in burned rangeland. In A. Busacca, S. Lilligren, and K. Newell (eds), *Dust Aerosols, Loess Soils and Global Change*, Washington State University, College of Agriculture and Home Economics, Pullman, WA, pp. 85-88.
- Tongway, D.J., and J. A. Ludwig. 1990. Vegetation and soil patterning in semi-arid mulga lands of eastern Australia. *Aust. J. of Ecol.* 15:23-34.
- Trumbore, S.E., O.A. Chadwick, and R. Amundson. 1996. Rapid exchange between soil carbon and atmospheric carbon dioxide driven by temperature change. *Sci.* 272:393-396.
- USDA-NRCS. 1998. *State of the Nation's Nonfederal Rangeland. 1992 NRI Summary-Nation, Region, and States*. USDA-NRCS. U.S. Government Printing Office. Washington, DC.
- USDI-BLM. 1998. *National Rangeland Inventory, Monitoring and Evaluation Report, Fiscal Year 1998*. BLM. USDI. U.S. Government Printing Office. Washington, DC.
- van Ginkel, J.H., and A. Gorrison. 1998. *In situ* decomposition of grass roots as affected by elevated atmospheric carbon dioxide. *Soil Sci. Soc. Am. J.* 62:951-958.
- van Ginkel, J.H., A. Gorrison, and J.A. van Veen. 1996. Long-term decomposition of grass roots as affected by elevated atmospheric carbon dioxide. *J. Environ. Qual.* 25:1122-1128.

- Vitousek, P.M., H.A. Mooney, J. Lubchenco, and J.M. Melillo. 1997. Human domination of earth's ecosystems. *Science* 277:494-499.
- Wedin, D.A., and D. Tilman. 1996. Influence of nitrogen loading and species composition on the carbon balance of grasslands. *Science* 274:1720-1723.
- Weltz, M., and M.K. Wood. 1986. Short-duration grazing in central New Mexico: effects on sediment production. *J. of Soil and Water Cons.* 41:262-266.
- Wilson, D.B. 1960. *Competition among three pasture species under different levels of soil nitrogen and light intensity*. Ph.D. Thesis. Oregon State University. Corvallis, OR.
- Wood, J.C., M.K. Wood, and J.M. Tromble. 1987. Important factors influencing water infiltration and sediment production on arid lands in New Mexico. *J. of Arid Environ.* 12:111-118.