

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

R.F. Follett • J.M. Kimble • R. Lal



LEWIS PUBLISHERS

Boca Raton London New York Washington, D.C.

CHAPTER 11

The Dynamics of Soil Carbon in Rangelands

G.E. Schuman,¹ J.E. Herrick,² and H.H. Janzen³

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.

¹Soil Scientist, Rangeland Resources Research, High Plains Grasslands Research Station, USDA, ARS, 8408 Hildreth Road, Cheyenne, WY 82009.

²Soil Scientist, Jornada Experimental Range, USDA, ARS, P.O. Box 3003, MSC 3JER, New Mexico State University, Las Cruces, NM 88003-8003.

³Research Scientist, Lethbridge Research Centre, Agriculture and Agri-Food Canada, P.O. Box 3000, Lethbridge, AB T1J 4B1 Canada.

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.

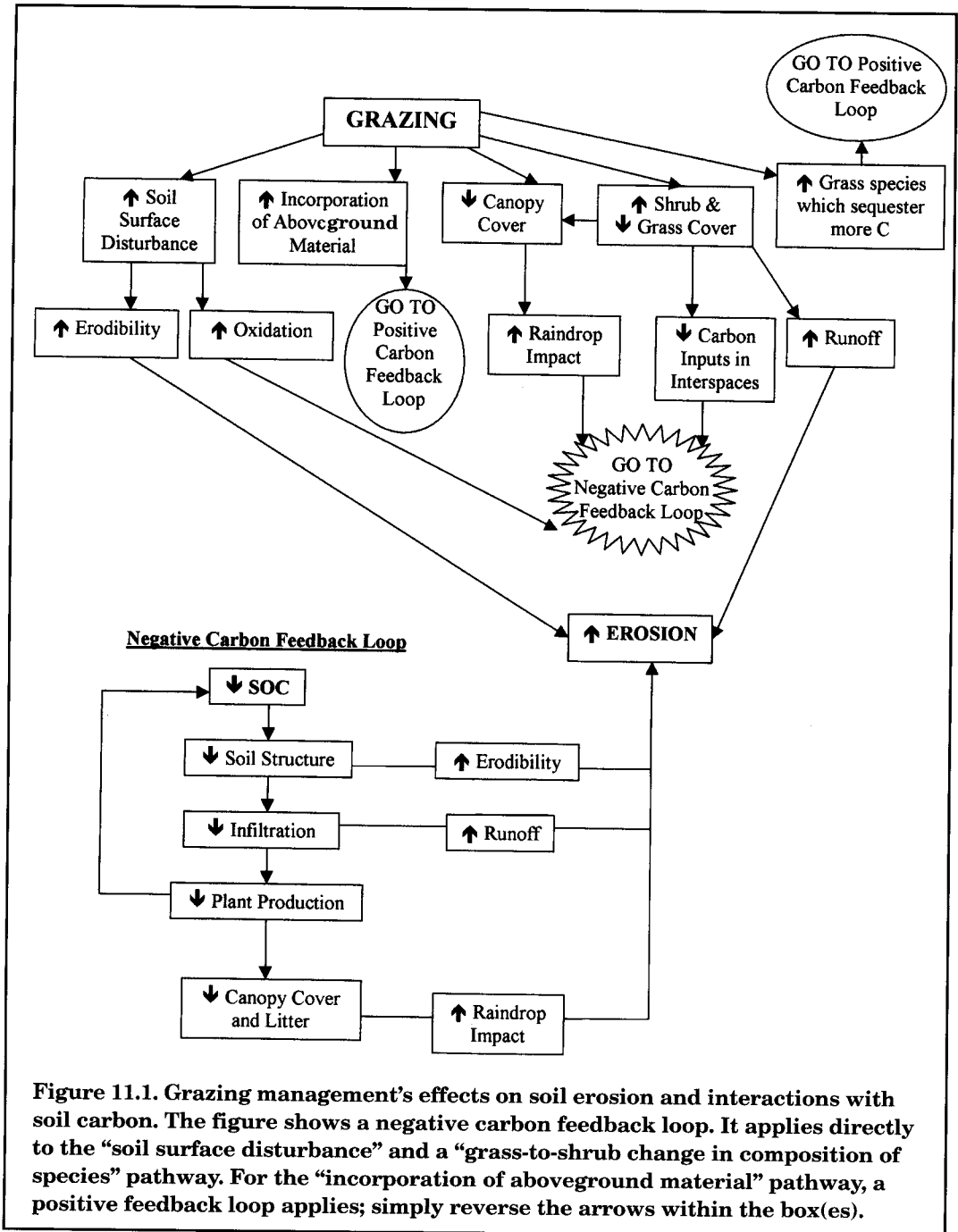


Figure 11.1. Grazing management's effects on soil erosion and interactions with soil carbon. The figure shows a negative carbon feedback loop. It applies directly to the "soil surface disturbance" and a "grass-to-shrub change in composition of species" pathway. For the "incorporation of aboveground material" 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-

