CHAPTER 6

Soil Carbon Dynamics and Rangeland Management

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CHAPTER OUTLINE

Abbreviations: CH_4, methane; CO_2, carbon dioxide; EPA, Environmental Protection Agency; GHGs, greenhouse gases; IPCC, International Panel on Climate Change; N_2O, nitrous oxide; SOC, soil organic carbon; SIC, soil inorganic carbon; U.S., United States

DEFINITION AND EXTENT OF U.S. RANGELANDS

Grazing lands occupy one-third of the total land base in the United States (U.S.), covering an estimated 316 to 336 Mha (Lal et al., 2003; USDA-ERS, 2006). Of this area, 80% is privately owned (214 to 257 Mha) and 20% is publicly owned (27 to 59 Mha) (Sobecki et al., 2001; USDA-NRCS, 2003) (Figure 6.1). Although a small proportion of grazing lands occur as grazed forest or hay land, most grazing lands in the conterminous U.S. occur west of the 95th meridian as rangelands (~80%), with the remainder occurring eastward as more intensively managed improved pastures (Mitchell, 2000; Schnabel et al., 2001). Over half of the total land base in the 17 western states of the conterminous U.S. is classified as rangeland. This chapter focuses on rangelands, which are defined as uncultivated lands managed with minimal inputs and consisting of extensively grazed native or naturalized plant species representative of historic climax vegetation (Figure 6.2; USDA-NRCS, 2003; Follett and Reed, 2010).

Rangelands represent one of the largest and most diverse land resources in the U.S., and encompass broad environmental gradients in temperature and precipitation (Liebig et al., 2012). Rangelands significantly impact both rural and national economies through the domestic livestock industry, with contributions assessed at $32 billion to the national economy in 2009 (USDA-ERS, 2010). A broad array of ecosystem goods and services, including

Managing Agricultural Greenhouse Gases. DOI: 10.1016/B978-0-12-386897-8.00006-1

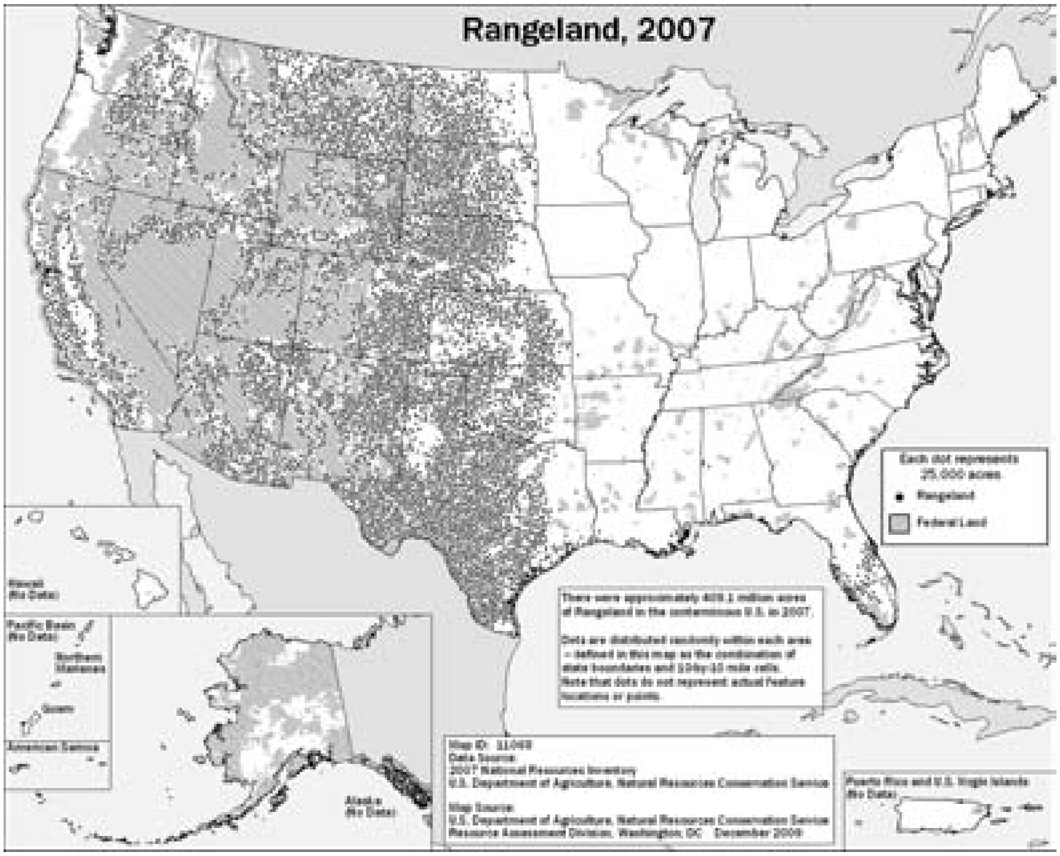

FIGURE 6.1
Extent of privately and publicly owned rangeland in the U.S. in 2007 (USDA-NRCS, 2009).

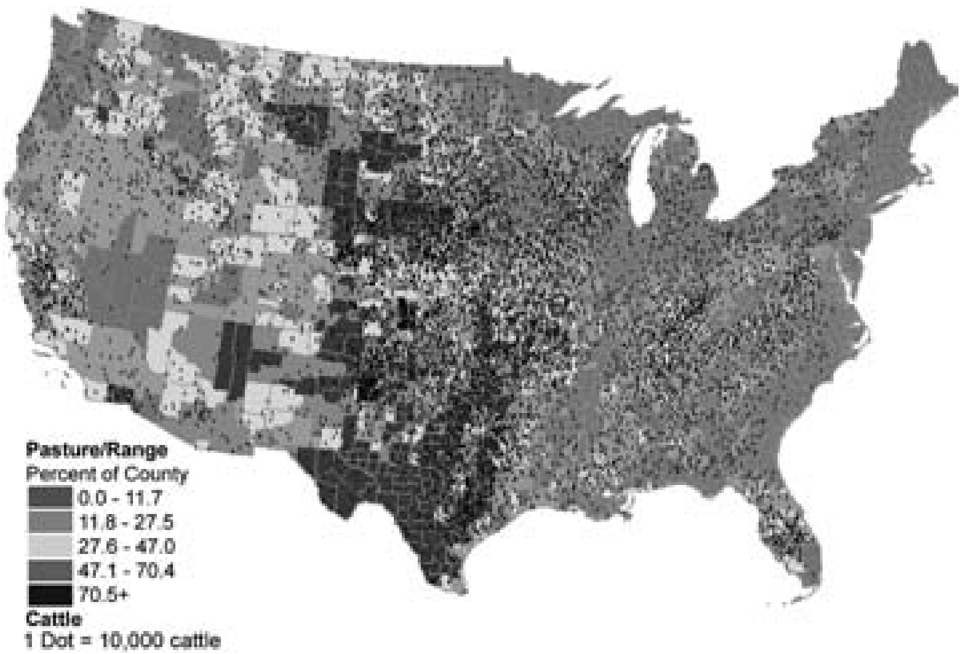

FIGURE 6.2
Extent of grazing land as a proportion of total county area and livestock production cattle inventory in the U.S. (one dot = 10,000 cattle; USGCRP, 2009). Please see color plate section at the back of the book.

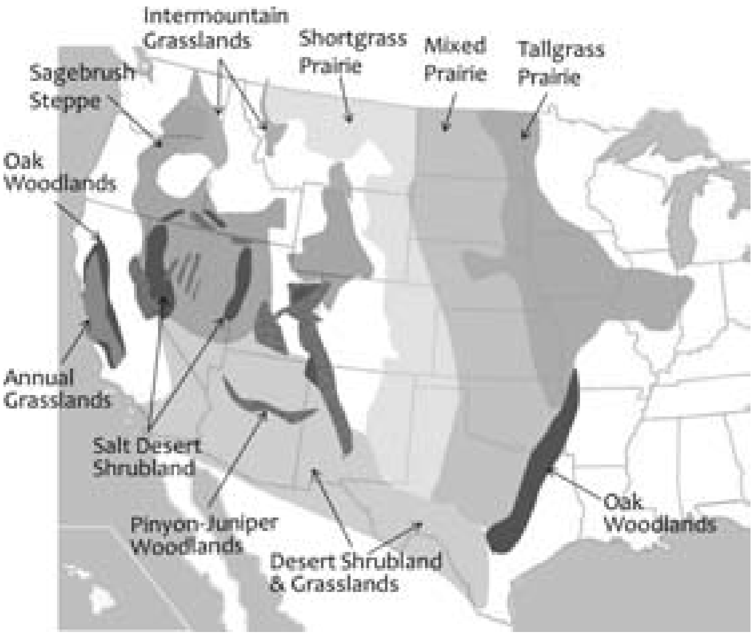

FIGURE 6.3
Major rangelands ecosystems of the U.S. (from Kuchler, 1964; courtesy of K. Launchbaugh). Please see color plate section at the back of the book.

livestock production, nutrient cycling, climate mediation, watershed function (flood control, storage, filtering), water quality and quantity, biodiversity, recreation, wildlife habitat, viewsheds, airsheds, and C sequestration, are provided from rangelands to society (Millennium Ecosystem Assessment, 2005; Havstad et al., 2007). Rangelands are characterized by a high diversity of climatic conditions and plant communities, including natural grasslands, savannas, most deserts, tundra, alpine plant communities, coastal and freshwater marshes, and wet meadows (USDA-NRCS, 2003). For classification purposes, rangelands are often separated into major ecosystems (Figure 6.3). Within any given rangeland ecosystem, landscapes can exhibit high diversity in plant communities, depending on topography. Further, plant community dynamics and associated ecological processes within these distinct ecological sites can be depicted by state transitions that occur between several different alternative plant community states in response to disturbance or management (Figure 6.4).

RANGELAND GHG MITIGATION POTENTIAL

Rangelands harbor considerable potential to mitigate climate impacts resulting from rising atmospheric levels of various greenhouse gases (GHGs), such as carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) due to an extensive land area (Schimel et al., 1990; Ojima et al., 1993; Conant et al., 2001), low-input management practices, and adaptive management facilitating active responses to highly variable within-year and between-year precipitation. Globally, rangelands could capture up to 2 to 4% of annual anthropogenic GHG emissions and up to 20% of the CO_2 released annually from global deforestation and land-use change (Derner and Schuman, 2007; Follett and Reed, 2010). Whether rangelands function as GHG sinks or sources will be determined by complex interactions between climate, vegetation, and grazing management and their effects on soils (both organic and inorganic C pools) and livestock (i.e. CH_4 emissions from enteric fermentation, N_2O emissions from manure) (Derner et al., 2006; Derner and Schuman, 2007; Ingram et al., 2008; Follett and Reed, 2010; Liebig et al., 2010a; Morgan et al., 2010). Rangelands are typically characterized by short periods of high C uptake during the growing season (2–3 months) and long periods of C balance or small losses during the remainder of the year (Svejcar et al., 2008), resulting in substantial interannual variability in net ecosystem exchange (Svejcar et al., 2008; Zhang et al., 2010).

The U.S. GHG emission inventory by the Environmental Protection Agency (EPA) groups emission estimates for six sectors defined by the International Panel on Climate Change

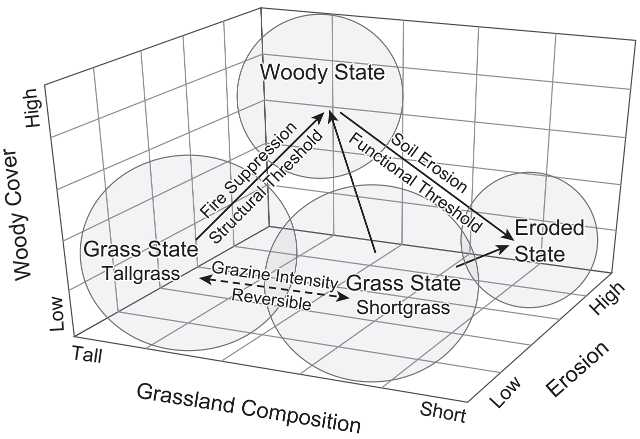

FIGURE 6.4
Example of a conceptual state-and-transition model showing selected alternative stable states that may occur on a given ecological site. Dashed and solid arrows show reversible and nonreversible transitions, respectively (from Briske et al., 2005).

(IPCC): Energy; Industrial Processes; Solvent Use; Agriculture; Land Use, Land-Use Change, and Forestry; and Waste. Emissions from livestock and rangelands are embedded within the Agriculture and Land Use, Land-Use Change, and Forestry sectors, respectively. Here, we report emissions estimates from the 2008 National GHG Inventory (EPA, 2011) from unconfined grazing systems for the following categories: enteric fermentation, manure management, and grasslands (inclusive of rangeland and improved pasture) (Table 6.1). In this inventory, grasslands on federal lands are included in the managed land base, but C stock changes are not estimated on these lands because federal grassland areas are assumed to have negligible changes in C due to limited land use and management change (EPA, 2011). Unconfined livestock values for enteric fermentation and manure management are for emissions from cattle, horses, sheep, and goats (Annex 3; EPA, 2011).

The dominant mechanisms for GHG mitigation in rangelands will be through C storage in soils and by minimization of livestock-related non-CO_2 emissions. For rangelands specifically, N_2O emissions from manure deposition is the primary GHG source (Liebig et al., 2012). Rangeland CH_4 fluxes from soil and areas of deposited manure are relatively small (~3% of total grazing land GHG emissions) due to predominantly aerobic conditions on most pastures

TABLE 6.1 Estimated 2008 Greenhouse Gases in Grasslands and Unconfined Livestock in the Conterminous US (EPA, 2011)[a]

	Livestock category (Tg CO_2 equivalents)				
Sources	*Cattle*	*Sheep*	*Goats*	*Horses*	*Total*
Enteric fermentation [CH_4][b]	89.1	1.0	0.3	3.6	94.0
Manure management [$CH_4 + N_2O$][b]	1.95	0.43	0.05	1.22	3.7
Grassland soil [direct + indirect N_2O]	–	–	–	–	61.7
Total greenhouse gas emissions	Unconfined livestock + grassland				159.4
	Total from agriculture[c]				427.5
	Total U.S.				6,957
Sinks	Grasslands remaining grasslands [CO_2]				(8.7)
	Lands converted to grasslands [CO_2]				(24.2)
Emissions mitigated (%)	Unconfined livestock + grassland emissions				21%
	Total agricultural emissions				8%
	Total U.S. emissions				0.5%

[a]*Totals may not sum due to independent rounding.*
[b]*Excluding contributions from confined livestock (beef cattle, dairy cattle, swine, poultry).*
[c]*Crop lands, grazing lands (including rangeland), and all livestock (confined + unconfined) emissions included; forest lands excluded.*

and ranges. In the conterminous U.S., modeled estimates of emissions from grasslands (including rangelands and improved pastures) and unconfined livestock accounted for approximately 37% of total agricultural GHG emissions in 2008 (Table 6.1) (EPA, 2011). Grasslands soils and vegetation combined, however, offset 21% of the total emissions estimated from grasslands and unconfined livestock.

Empirical studies measuring the net effects of GHG emissions from rangelands, however, are limited (Liebig et al., 2012). Direct measurements suggest that grazing management has minimal effects on rangeland N_2O emissions (Liebig et al., 2010a; Wolf et al., 2010). Similar to IPCC results, measured CH_4 emissions from the soil are minimal in grazing systems (Liebig et al., 2010a), with the dominant source being CH_4 emitted from livestock enteric fermentation. Although animal density on the land will control livestock-related CH_4 emissions per unit area, simply transferring those animals to another part of the rangeland landscape is not likely to result in a general reduction of CH_4 emissions from this ecosystem.

Rangeland C sequestration rates on a per unit area basis are relatively low relative to croplands, improved pastures, or forested lands, but can still have a substantial impact on overall GHG emissions because of their large land base. For specific categories within rangelands (i.e. soils, vegetation, livestock), potential mitigation or emission pathways are not well defined (Figure 6.5). While much research has focused on rangeland soil organic carbon (SOC) pools, little is known about alternative pathways for soil C storage or emission, such as soil accumulation of pyrogenic C derived from burning (i.e. charcoal, or black C) or CO_2 fluxes from the soil inorganic C (SIC) pool. For rangeland vegetation, management and local climate condition affect plant community dynamics and subsequent plant C inputs into soils (Derner et al., 2006; Ingram et al., 2008; Zhang et al., 2010). Two factors, however, have emerged as the primary controls on the fate of SOC in rangelands: (1) plant productivity (i.e. above- and belowground biomass quantity, plant nutrient quality) (Derner et al., 2006; Derner and Hart, 2007; Piñeiro et al., 2010); and (2) direct and indirect effects of grazing on vegetation composition (Derner and Schuman, 2007; Bagchi and Ritchie, 2010) (Figure 6.5). Management of rangeland plant communities will also be impacted by state changes after disturbances such as fire, plant invasions (i.e. non-native annuals, woody plants) (Figure 6.4), or managed introductions of desirable species (i.e. N-fixing legumes). Finally, non-CO_2 GHG emissions from unconfined livestock will be affected most strongly by stocking rate. Stocking rate also controls C and nutrient inputs from manure into soils as well as the level of physical disturbance related to hoof action or overgrazing, which can lead to increased wind and water

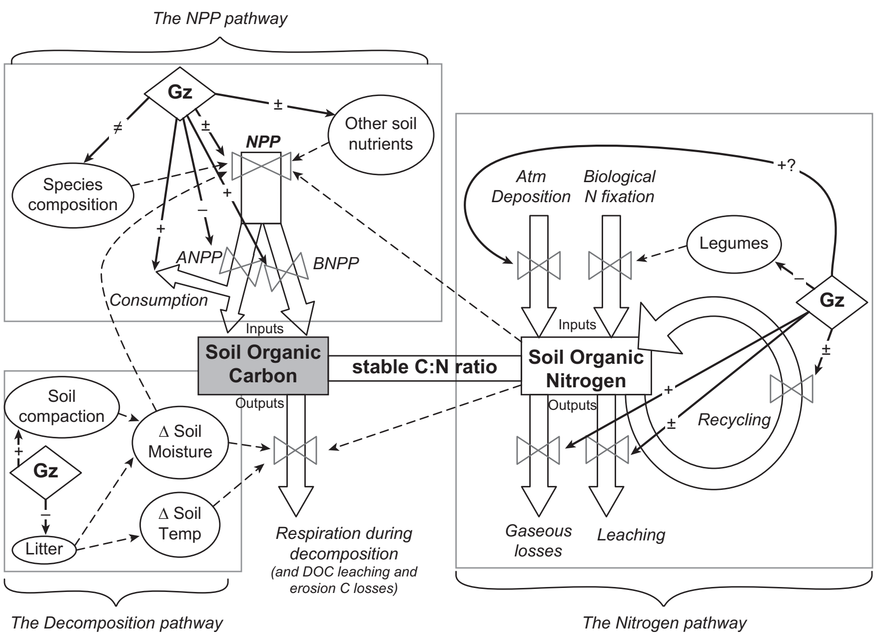

FIGURE 6.5
Effects of grazing on soil organic carbon pools via changes in net primary production (NPP), nitrogen stocks, nitrogen or cycling, and/or decomposition. ANPP, aboveground NPP; BNPP, belowground NPP; C:N, carbon-to-nitrogen ratio; DOC, dissolved organic carbon; Gz, grazing (from Piñeiro et al., 2010).

erosion of rangeland soils. The GHG impacts of changes in vegetation species composition and supplemental feeding practices on livestock emissions have focused primarily on C dynamics, while the full complement of GHG emissions have rarely been addressed (Liebig et al., 2005, 2010a). Long-term management for the sustainable use of rangelands will depend ultimately on elucidating the mechanisms driving rangeland processes to improve our ability to predict ecosystem-level responses to projected changes in temperature and precipitation.

RANGELAND C DISTRIBUTION: VEGETATION AND SOILS

The distribution of C between vegetation and various soil pools varies greatly across U.S. rangelands, reflecting the tremendous variability in climate and plant community types (Figure 6.3). Plant aboveground biomass can store <5% of total ecosystem C in non-woody plant dominated ecosystems (Derner et al., 2006) to >25% in pinyon-juniper dominated ecosystems (Rau et al., 2010). Soils contain the majority of the C in rangelands. Total soil C (organic plus inorganic) in rangeland soils to 1 m depth range from 90 to 266 Mg C ha^{-1} (Guo et al., 2006; Figure 6.6A, B). Soil organic C concentration of rangeland soils varies from <1 to >10% (Janzen, 2001). Rangelands tend to be C sinks (Schlesinger, 1997; USDA, 2008), with SOC sequestration rates of up to 0.5 Mg C ha^{-1} yr^{-1} (Schuman et al., 1999; Derner and Schuman, 2007; Morgan et al., 2010; Liebig et al., 2010a).

Greater than 90% of stored C will be sequestered in the soil as SOC derived from decomposing plant matter and microbial biomass, but two other potential pathways for C sequestration are: (1) the formation of charcoal (i.e. black C) which is stored in the SOC pool, and (2) the formation of secondary carbonates which are stored in the soil inorganic C (SIC) pool. Vegetation composition in U.S. rangelands is commonly controlled using prescribed fires, particularly in preventing or slowing the invasion of woody plant encroachment (Scholes and Archer, 1997). In the southern Great Plains, two to three prescribed fire events can significantly increase soil black C content (Dai et al., 2005). Black C produced from burning aboveground biomass results in extremely stable forms of organic C (i.e. highly aromatic, few functional groups), with mean residence times of >10,000 years (Kuhlbusch, 1998; Swift, 2001). In rangeland soils worldwide, black C can make up 5 to 35% of the total SOC pool (Skjemstad et al., 1999; Glaser and Amelung, 2003; Dai et al., 2005). Its small particle size (<53 μm) also makes black C relatively mobile, resulting in long-term storage of this highly resistant C deeper in the soil profile (Skjemstad, 1999; Dai et al., 2005).

The uptake or release of CO_2 from SIC pools could also contribute to the role of rangeland soil as a C source or sink. In contrast to the rest of the U.S., a vast pool of SIC occurs as carbonates in semiarid and arid rangeland soils (Monger and Martinez-Rios, 2001; Emmerich, 2003; Guo et al., 2006) (Figure 6.6A, B). The prevalence of SIC in rangelands is promoted by the combination of more xeric conditions and the predominance of calcareous soil parent materials, contributing to calcification and secondary carbonate formation (Batjes and Sombroek, 1997). In various Land Resource Regions (LRR; USDA-NRCS) encompassing rangeland and/or grazing land (Figure 6.7), SIC can equal or exceed SOC stores in the top 2 meters of the soil profile (Guo et al., 2006) (Table 6.2). Inorganic C storage or loss pathways, however, are poorly understood (Follett et al., 2001; Liebig et al., 2006; Svejcar et al., 2008), which is reflected in the wide range of predicted SIC sequestration values (0.0012–0.120 Mg C ha^{-1} yr^{-1}; Gile et al., 1981; Reheis et al., 1992, 1995; Schlesinger, 1997).

RANGELAND MANAGEMENT IMPACTS

Management practices that maximize soil C storage while minimizing both direct and indirect emissions of non-CO_2 GHGs from grazing livestock will have the greatest impacts on rangeland GHG mitigation potential. Range management has long recognized vegetation as the pivotal management component affecting soil quality (i.e. quantity and quality of above- and

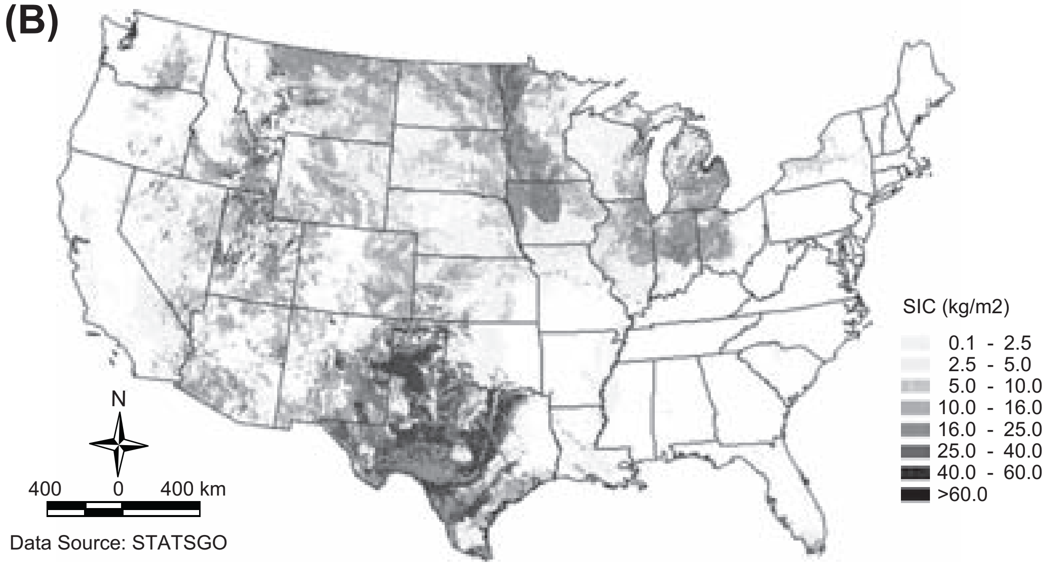

FIGURE 6.6
Distribution of (A) soil organic C (SOC) and (B) soil inorganic C (SIC) to 2 m soil depth in the conterminous U.S. (from Guo et al., 2006). Please see color plate section at the back of the book.

belowground C inputs), livestock productivity (i.e. quantity, quality, and timing of available forage), and overall ecosystem sustainability (i.e. productivity of plant and livestock, soil erosion control, resistance to invasive plant species and subsequent impacts on altered fire regime). Range management approaches have evolved over time from linear Clementsian succession-based concepts towards more complex state-and-transition models that better incorporate disturbance, climate, and multi-directional changes in plant community dynamics (Westoby, 1980; Young and Clements, 2001; Briske et al., 2005). Rates of SOC sequestration decrease with longevity of a management practice (Derner and Schuman, 2007), indicating that ecosystems reach a "steady state" and that changes in management or inputs may be required to sequester additional C (Conant et al., 2001, 2003; Swift, 2001). Key considerations for SOC sequestration in rangelands are: (1) the aboveground C pool is a minor component of total ecosystem C storage and mean residence time of this C pool is only a few years, so yearly variations in aboveground biomass minimally affect C storage; (2) most SOC is recalcitrant

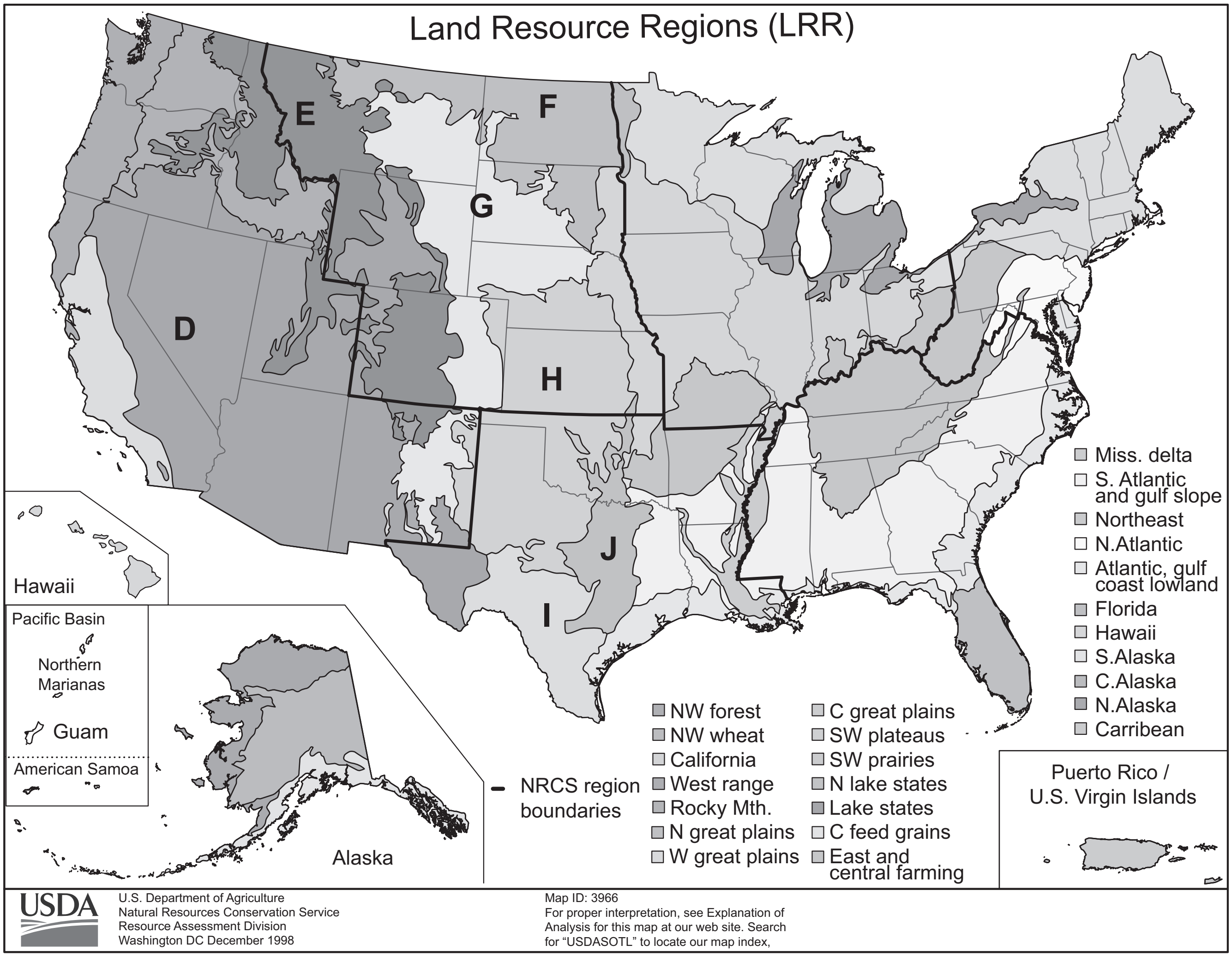

FIGURE 6.7
Land Resource Regions (LRRs) in the U.S. (USDA-NRCS). LRRs in Table 6.2 are labeled. Please see color plate section at the back of the book.

TABLE 6.2 Mean (±1 Standard Deviation) Soil Organic C (SOC) and Soil Inorganic C (SIC) Storage in Western Land Resources Regions (LRRs) that Include Grazing Lands and/or Rangelands (adapted from Guo et al., 2006)

LRR	Region description	Area	SOC (Mg C ha^{-1})		SIC (Mg C ha^{-1})	
		Mha	*0–20 cm*	*0–100 cm*	*0–20 cm*	*0–100 cm*
D	Western range and irrigated	127	14 ± 16	35 ± 52	11 ± 24	57 ± 112
E	Rocky Mountain range and irrigated	52	28 ± 27	66 ± 88	3 ± 11	24 ± 78
F	Northern Great Plains spring wheat	35	50 ± 29	117 ± 72	5 ± 12	73 ± 100
G	Western Great Plains range and irrigated	52	20 ± 12	50 ± 30	7 ± 17	46 ± 83
H	Central Great Plains winter wheat and range	58	27 ± 12	80 ± 42	8 ± 20	65 ± 134
I	Southwest plateaus and plains range and cotton	17	24 ± 10	71 ± 62	45 ± 48	195 ± 219
J	Southwestern prairies cotton and forage	14	24 ± 12	81 ± 60	28 ± 46	155 ± 242

and well protected from minor natural disturbances; microbial biomass and particulate or light-fraction organic C are most sensitive to management or land-use change, whereas chemically resistant organic C and soil carbonates are least sensitive (Allen et al., 2010); (3) major pathways of SOC accumulation include rhizodeposition belowground, surface deposition of animal feces, decaying litter from aboveground vegetation, and the activity of soil biota (bacteria, fungi, protists, and fauna) promoting humic substance synthesis, aggregate formation, and subsequent long-term SOC stabilization (Six et al., 2002; Jones and Donnelly, 2004; Ingram et al., 2008, Piñerio et al., 2010); and (4) large perturbations in the SOC pool can occur with major soil disturbances (i.e. tillage, wind and water erosion, surface denudation with overgrazing). These impacts can occur naturally with extreme weather conditions (i.e. drought) or through poor management decisions that reduce the vigor of plant communities (Follett et al., 2001).

Grazing Management

Improved grazing management is central to rangeland health (Derner and Schuman, 2007; Follett and Reed, 2010; Morgan et al., 2010). Improved management strategies include using appropriate stocking rate and forage utilization, timing grazing to avoid the months of high C uptake and adjusting the frequency of grazing (i.e. destocking during drought conditions), and implementing adaptive management practices to promote active responses to highly variable within-year and between-year precipitation. Shifts in plant community composition due to grazing can influence SOC. For example, SOC was greater in grazed compared to ungrazed areas in northern mixed grass prairie (Frank et al., 1995; Manley et al., 1995; Schuman et al., 1999; Reeder and Schuman, 2002; Ganjegunte et al., 2005, Liebig et al., 2006, 2010a), partially due to the increased dominance of the grazing-resistant species blue grama (*Bouteloua gracilis*) which produces greater root biomass in the upper soil profile compared to the midgrass species it replaces under grazing (Derner et al., 2006).

Rangeland SOC response to stocking rate and grazing intensity is variable (Smoliak et al., 1972; Wood and Blackburn, 1984; Warren et al., 1986; Biondini et al., 1998; Schuman et al., 1999; Liebig et al., 2006, 2010a). Observed changes in SOC due to grazing have been attributed to differential grazing impacts across environmental gradients (Derner et al., 2006) as well as to management-by-climate interactions (e.g. drought) (Ingram et al., 2008). For example, heavy grazing alone on restored rangelands can slow down the rate of soil C accumulation (Fuhlendorf et al., 2002), but heavy grazing during drought conditions can result in overall losses in SOC (Schnabel et al., 2001; Ingram et al., 2008).

The response of SOC to a specific grazing method has been investigated sparsely, at best. Two studies suggest an increase in SOC with rotational grazing compared with continuous season-long grazing (Conant et al., 2003; Teague et al., 2010), but another study found no differences in SOC between these grazing systems (Manley et al., 1995). The majority of available scientific evidence suggests that rotational grazing has no direct impact on broad-scale vegetation production or composition (Briske et al., 2008). Any change in SOC under rotational grazing, therefore, is expected to reflect vegetation changes that are independent of stocking rate.

N-fixing Legumes

Many rangelands are N-limited, so increasing N inputs by interseeding N-fixing legumes can increase forage production and quality as well as C sequestration (Mortenson et al., 2004, 2005; Liebig et al. 2010b). The introduction of N-fixing legumes as an alternative to N-fertilization has been the subject of research for decades (Tesar and Jakobs 1972; Heinrichs 1975; Kruger and Vigil 1979; Berdahl et al., 1989), providing critical insights to both shorter- and longer-term management impacts of interseeding legumes in rangelands. At one long-term study site in northern mixed-grass prairie, interseeding yellow-flowered alfalfa (*Medicago sativa* ssp. *falcata*) increased soil C by 4—17% across three interseeding dates (Mortenson et al., 2004), corresponding to 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. In addition, interseeding significantly increased soil total N, aboveground production, and N forage quality (Mortenson et al., 2004, 2005) did not impact soil N_2O emissions (Liebig et al., 2012).

Prescribed Fire

Burning can indirectly contribute to rangeland C sequestration by enhancing nutrient availability for the following season's plant growth, which can exceed C combustion losses (Swift, 2001). Burning, therefore, indirectly affects photosynthesis (Knapp, 1985; Svejcar and Browning, 1988; Bremmer and Ham, 2010), soil and canopy respiration (Knapp et al., 1998; Bremmer and Ham, 2010), and can alter species composition (Pacala et al., 2007; Boutton et al., 2009). Carbon losses from burning herbaceous plant-dominated grazing lands is a minor component of the total annual ecosystem C emitted (Owensby et al., 2006; Bremmer and Ham, 2010). Burning rangelands with a significant above ground woody component, however, can result in immediate substantial ecosystem C losses (Rau et al., 2010). Further, C losses in western U.S. rangelands via burning of woody biomass could be compounded by additional losses of surface soil SOC derived from historical (i.e. >100 years) woody plant encroachment (Boutton et al., 2009; Neff et al., 2009).

INTERACTIONS BETWEEN MANAGEMENT AND CLIMATE

Climate and management can influence soil C and GHG emissions on rangelands (Schuman et al., 1999; Follett et al., 2001; Jones and Donnelly, 2004; Derner et al., 2006; Derner and Schuman, 2007; Ingram et al., 2008; Svejcar et al., 2008; Bremmer and Ham, 2010; Liebig et al., 2010a; Piñeiro et al., 2010; Rau et al., 2010; Zhang et al., 2010). Specifically, both short-term weather conditions (e.g. droughts) (Ciais et al., 2005; Soussana and Lüschert, 2007; Ingram et al., 2008; Svejcar et al., 2008; Zhang et al., 2010) and long-term changes in the global environment (e.g. rising temperature, altered precipitation patterns, rising atmospheric CO_2 concentrations) can reduce soil quality by altering plant and microbial community compositions (Jin and Evans, 2010; Jin et al., 2011) and changing forage quality (Milchunas et al., 2005; Soussana and Lüschert, 2007; Hatfield et al., 2008; Morgan et al., 2008, 2010). Adaptive management practices that optimize rangeland vegetation and soil responses to changing environmental conditions, therefore, play a significant role in determining whether rangelands will lose or sequester C (Ingram et al., 2008; Svejcar et al., 2008).

Projected increases in temperature due to climate change may constrain rangeland C storage or cause rangelands to become C sources, particularly in drought-prone regions. For example, in the Great Plains region under non-drought conditions, rangelands to the west are generally C sources and eastern rangelands are C sinks. In years when drought affects >65% of the region, however, the Great Plains becomes an overall C source (Zhang et al., 2010). In contrast, the interaction of warmer temperatures with rising atmospheric CO_2 over the past century has enhanced net primary production (Hatfield et al., 2008) which could increase rangeland C sequestration. Such positive effects of climate change on productivity are expected at northern latitudes and at high altitudes where temperature is an important factor limiting production and annual precipitation is not expected to decline. Increased C inputs due to CO_2-enhanced plant growth, however, can also stimulate microbial decomposition of older SOC pools in the short term (i.e. priming effect; Kuzyakov 2002) causing soils to become C sources (Carney et al., 2007). If warming accelerates, the CO_2 benefit to plant productivity and water use efficiency could be offset further by increased desiccation stresses that constrain or reduce plant productivity, especially in the southwestern quadrant of North America (Seager et al., 2007). Other phenomena with important implications for GHG mitigation, such as woody plant encroachment (e.g. Boutton et al., 2009; Neff et al., 2009), likely have been derived from several of these factors (Morgan et al., 2008, 2010; Van Auken, 2009).

KNOWLEDGE GAPS AND FUTURE RESEARCH NEEDS

Knowledge gaps, both research and informational, exist for mitigation of GHGs in rangelands (Derner and Schuman, 2007; Morgan et al., 2010). First, from the research perspective, almost all the prior experimental efforts have focused on individual GHGs, particularly CO_2. Emissions of CH_4 and N_2O, however, might counteract SOC sequestration and lead to positive or negative GHG balances (Liebig et al., 2010a), but such studies of total GHG budgets are lacking. Second, very limited research efforts have addressed mitigation of GHGs in arid rangelands, particularly shrublands. Third, interactions between management and environment on GHGs in rangelands have revealed interesting preliminary findings (Ingram et al., 2008), but further quantification is needed for more robust interpretations and guidelines for recommendations. Fourth, adaptive management practices that more closely match forage demand with forage availability in highly variable environments (i.e. high intra- and inter-annual precipitation variability) are based largely on experiential knowledge rather than empirical evidence. Finally, simulation models used in broad national inventory estimates may not be appropriate for smaller-scale, enterprise-level (i.e. individual ranch) GHG accounting, complicating how conservation practices are selected for targeting land areas that provide the largest potential returns to the general public.

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