



United States  
Department  
of Agriculture

Forest Service

**Rocky Mountain  
Research Station**

General Technical  
Report  
RMRS-GTR-316

February 2014



# Review of Climate Change Impacts on Future Carbon Stores and Management of Warm Deserts of the United States

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## Abstract

Reducing atmospheric carbon dioxide (CO<sub>2</sub>) concentration through enhanced terrestrial carbon storage may help slow or reverse the rate of global climate change. As a result, Federal land management agencies, such as the U.S. Department of Agriculture Forest Service and U.S. Department of the Interior Bureau of Land Management, are implementing management policies to increase carbon storage. However, information on how projected southwestern climate changes might affect the balance between CO<sub>2</sub> uptake and loss on semiarid rangelands is not easily accessible to land managers. We summarize studies that focus on key components of carbon exchange, including photosynthesis, soil respiration, and plant productivity, across the warm deserts of North America to determine if common trends exist that can be utilized in management. We also provide an overview of how management practices can influence carbon sequestration in this region and discuss the U.S. Department of Agriculture Forest Service Climate Change Scorecard. Since desertification is projected to increase in the future, management strategies that increase carbon sequestration or decrease carbon loss are especially important. This requires managers to thoughtfully consider management practices that do not impede sequestration during critical times. For a popular version of the GTR see Rangelands February 2014.

**Keywords:** Global climate change; carbon dioxide (CO<sub>2</sub>); carbon sequestration in arid-semiarid Ecosystems; Mojave, Sonoran and Chihuahuan Deserts; land management to increase carbon storage

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## Acknowledgments

Funding in support of this research was provided by a grant from the USDA Forest Service, Western Wildland Environmental Threat Assessment Center. This research was completed in partial fulfillment of the requirements for the degree of Doctor of Philosophy from the University of New Mexico. We thank Jesse Nippert for providing comments to the dissertation chapter; Sandra Brantley, Rosemary Pendleton, Linda Joyce, and Carly Woodlief for comments or reviews; and Yancey Ranspot for technical help. We thank Dr. Zhiliang Zhu and his LANDCARBON team for enabling us to use data for Figure 2.

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# Review of Climate Change Impacts on Future Carbon Stores and Management of Warm Deserts of the United States

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## Introduction

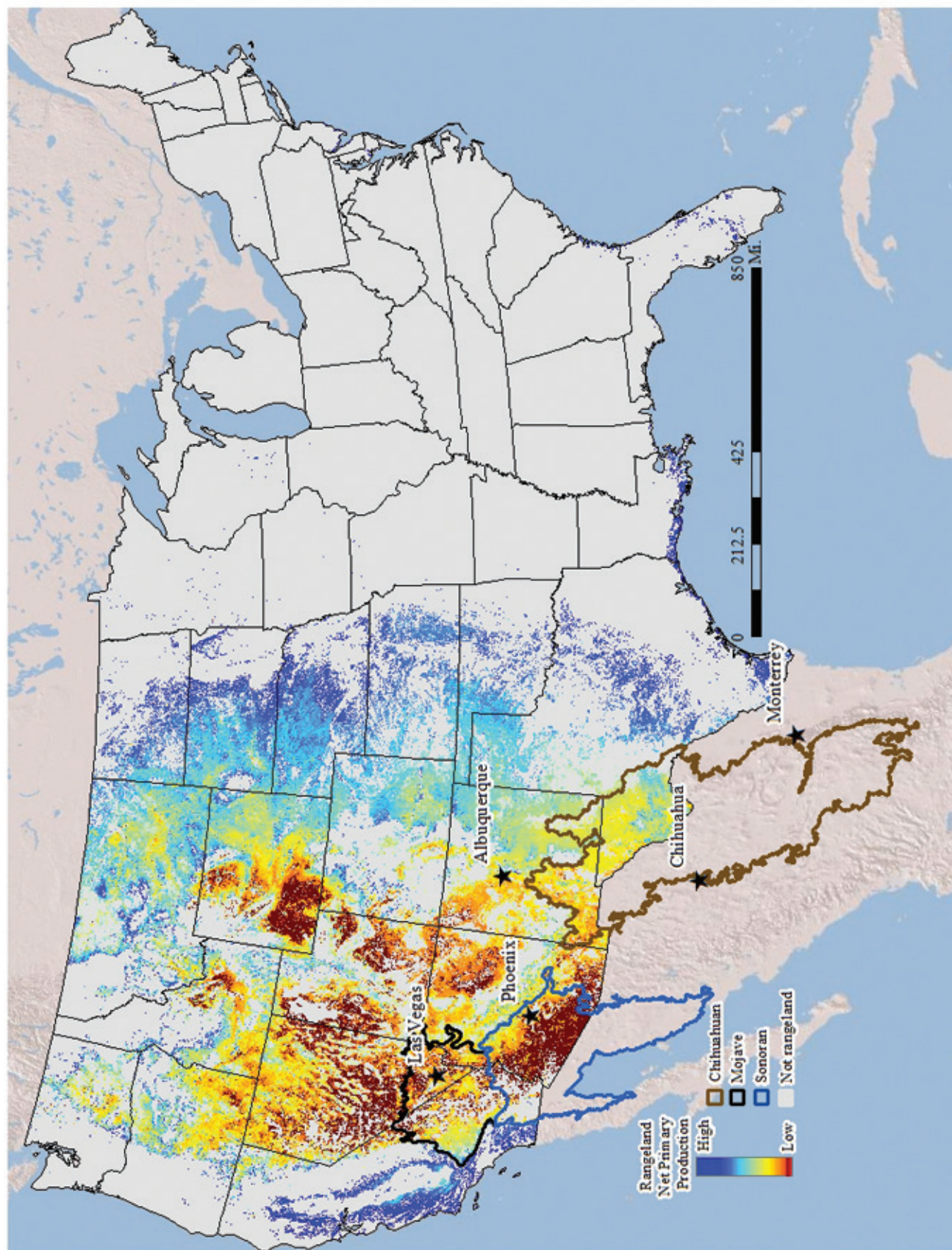
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In the terrestrial carbon cycle, CO<sub>2</sub> acquired from the atmosphere through photosynthesis is assimilated in plants (GPP, Appendix 1) while CO<sub>2</sub> is also emitted back to the atmosphere as the byproduct of autotrophic and heterotrophic respiration (hereafter, R<sub>eco</sub>) (Kondratyev and others 2003; Adams 2009). Together, these metabolic processes serve as the engine that drives the terrestrial carbon cycle (Chapin and others 2002).

Because of fossil CO<sub>2</sub> emissions, ecosystems are subject to shifts in climate patterns (Karl and others 2009; Allison and others 2011). While the degree of impact may vary by latitude, increasing CO<sub>2</sub> concentrations, warmer temperatures, and changes in the frequency of extreme weather and climate events will alter the balance between GPP (net CO<sub>2</sub> uptake) and R<sub>eco</sub> (CO<sub>2</sub> loss) or net ecosystem exchange (NEE) and will influence the amount of CO<sub>2</sub> remaining in the atmosphere (Adams 2009).

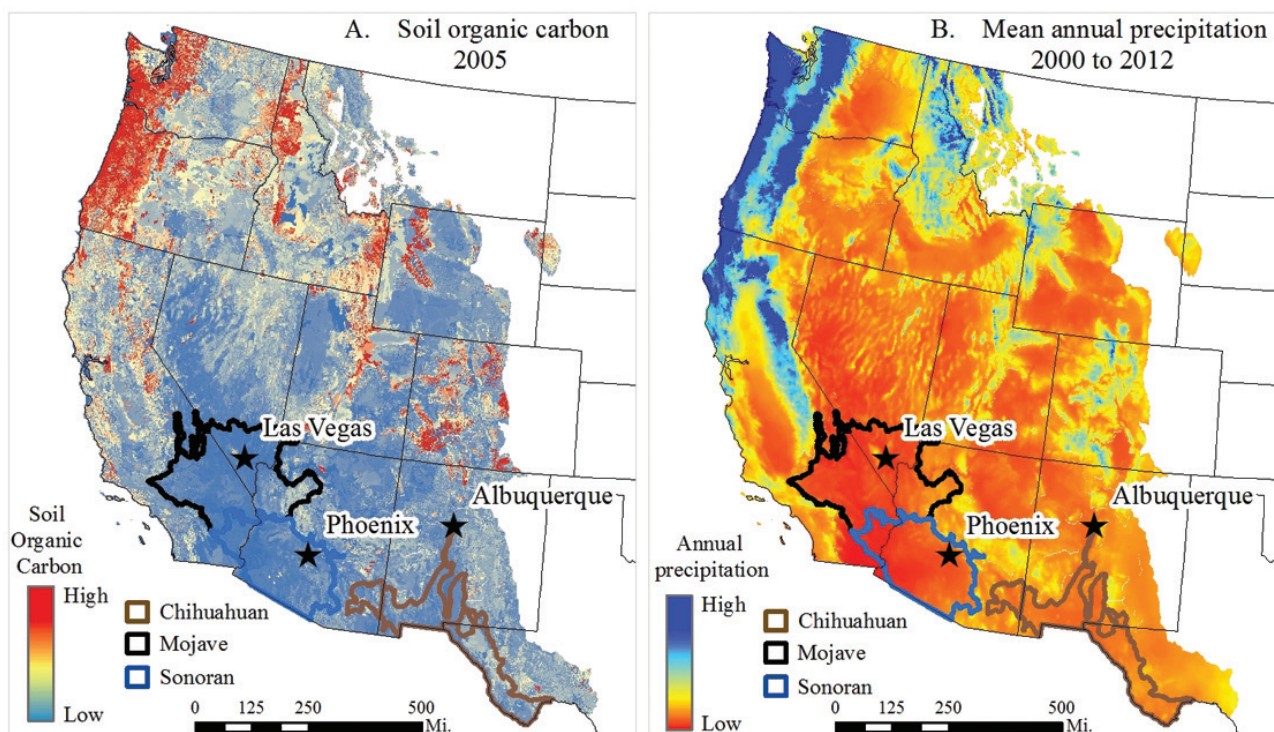
Although arid-semiarid regions are characterized by low primary productivity (Figure 1), they cover ~45% of the land surface (Schimel 2010) and, therefore, collectively contribute significantly to the global carbon cycle. This biome stores ~199 Pg C in vegetation and in soil organic carbon (Janzen 2004). Moreover, the soils also contain inorganic carbon as carbonate (CaCO<sub>3</sub>), which sequesters an additional 800-1700 Pg C (Schlesinger and others 2000). For this reason, arid-semiarid soils are considered the third largest global pool of carbon (Emmerich 2003). In these water-limited ecosystems, CO<sub>2</sub> loss through R<sub>eco</sub> frequently exceeds GPP as net CO<sub>2</sub> uptake is limited to periods of favorable rainfall (Anderson-Teixeira and others 2011). Arid-semiarid ecosystems are highly responsive to precipitation variability and, therefore, will be affected by predicted increases in extreme precipitation events, increased CO<sub>2</sub> concentrations, and increased temperatures (Smith and others 1997; Heisler-White and others 2008; Diffenbaugh and others 2008). Therefore, it is important to understand how these elements of climate change might affect NEE (Figure 2).

In this review, we summarize the literature from the warm deserts of North America and consider leaf (A<sub>net</sub>), soil (Rs), and plant community (aboveground net primary productivity; ANPP) studies from the Mojave, Sonoran, and Chihuahuan Deserts to determine if common trends exist across measurement scales that can be utilized to effectively manage the warm deserts. Furthermore, we investigate the potential management strategies that may increase carbon sequestration or minimize carbon losses in this region. We begin by describing the warm deserts in the southwestern United States and consider climate change projections for this region. We then examine how climate change may affect NEE and carbon sequestration.



**Figure 1**—Rangeland net primary production estimated from the Moderate Resolution Imaging Spectroradiometer (MODIS) NPP Collection 4.5 product (Running and others 2004). Note the predominantly low net primary production that characterizes deserts.





**Figure 2**—Soil organic carbon adapted from the LANDCARBON Project (Zhu and Reed [2012]). Note the relationship between mean annual precipitation and soil organic carbon. Generally speaking, greater precipitation enables greater rates of carbon accumulation. Mean annual precipitation was generated from the Parameter-elevation Regressions on Independent Slopes Model (PRISM) data (Daly and others [2001]) (<http://www.prism.oregonstate.edu/>). Deserts exhibit both low annual precipitation and low carbon storage ability relative to other biomes.

## The Deserts of North America

The North American Desert biome is divided into cold and warm deserts. The cold desert, including the Great Basin and Colorado Plateau, is primarily located in Nevada but also extends into portions of Utah, Colorado, Arizona, New Mexico, Oregon, Washington, Idaho, and Wyoming (Barbour and Billings 2000). Compared to warm deserts, cold deserts: occur at higher elevations, have more days with temperatures that reach freezing, receive precipitation that mostly falls as snow during the winter, and maintain deeper soil water recharge due to lower evapotranspiration and the slow release of water through snowmelt (Smith and others 1997). The warm deserts are divided into three types: (1) Mojave (Figure 3), (2) Sonoran (Figure 4), and (3) Chihuahuan (Figure 5). While the largest area of the Sonoran and Chihuahuan Deserts is located in Mexico (Figure 1), in the United States these desert types are distributed from Nevada into southern California (Mojave) and east through southern Arizona (Sonoran), southern New Mexico (Chihuahuan), and western Texas (Chihuahuan). We limit the remainder of our discussion to the warm desert types.





**Figure 3**—Mojave Desert. Photo credit: Ken Cole, U.S. Geological Survey.



**Figure 4**—Sonoran Desert. Photo credit: U.S. Fish and Wildlife Service.





**Figure 5**—Chihuahuan Desert. Photo credit: Sevilleta LTER Program.

A common trait of the Mojave, Sonoran, and Chihuahuan Deserts is that they are located on the leeward (e.g., rain shadow) side of the Sierra Nevada, Peninsular and Sierra Madre Occidental, and Sierra Madre Oriental mountain ranges, respectively. Another shared trait is that precipitation is spatially and temporally variable and occurs in discrete packages (i.e., “pulses”) that initiate biotic activity followed by a dry inter-pulse period when the ecosystem returns to a less biologically active state (Noy-Meir 1973). Finally, in the southwestern United States, short-term variability in winter rainfall derived from the Pacific is largely influenced by the El Niño and La Niña phases of the Southern Oscillation (Merideth 2001). The El Niño phase is largely associated with the high pressure system over Darwin, Australia, that temporarily reverses position with the low pressure system located over Tahiti (Barry and Chorley 2003). This exchange of pressure systems results in the breakdown of the trade winds, warmer (SST) along the coast of South America, and a low pressure zone of rising air and increased precipitation, leading to above-average rainfall across the warm deserts (Merideth 2001). In contrast, the La Niña phase is characterized by a strengthening of the trade winds that bring below-normal SST along the coast of South America and above-normal SST to the western Pacific Ocean (Duxbury and Duxbury 1993). This shift in SST results in the northward displacement of the subtropical and mid-latitude jet streams, leading to dry winters (Duxbury and Duxbury 1993; Barry and Chorley 2003) across the warm deserts. Beyond these similarities, the warm deserts are distinguished by climatic differences. Two main differences are: (1) the precipitation gradient with rainfall increasing from the

Mojave Desert east to the Chihuahuan Desert, and (2) the seasonality of precipitation with a winter dominant rainfall in the Mojave, summer and winter dominant rainfall (bimodal) in the Sonoran, and summer dominant rainfall in the Chihuahuan Desert with some winter precipitation inputs.

The Mojave Desert is the driest of the warm deserts. The mean annual temperature range is from 1.5 to 28.7 °C and winter snowfall can occur (Beatley 1974; Rundel and Gibson 1996). Average rainfall is 137-168 mm/year and is characterized by a winter dominant rainfall pattern in which most rainfall occurs from October through April as a result of west-to-east frontal systems originating over the Pacific Ocean (Smith and others 1997; Hereford and others 2006). This rainfall pattern is particularly important for the C<sub>3</sub> perennial shrubs (*Larrea tridentata*, *Ambrosia dumosa*, *Atriplex hymenelytra*, *A. polycarpa*, and *Coleogyne ramosissima*) that dominate the landscape (Beatley 1974; Rundel and Gibson 1996; MacMahon 2000). The eastern margins of the Mojave can experience a winter/summer (e.g., bimodal) rainfall pattern when the NAMS (North American Monsoon System) expands into the region (Rundel and Gibson 1996). For example, Hereford and others (2006) found a winter dominant rainfall pattern at 90% of weather stations located west of 117° W longitude (Barstow, California) while a bimodal rainfall pattern occurred at 70% of the weather stations located east of this longitude. The eastern region is also referred to as the Colorado Desert and is considered to be an extension of the Sonoran Desert (Schoenherr 1992).

The Sonoran Desert is the warmest of the desert types and is defined as subtropical in nature (Rundel and Gibson 1996). Because this desert type rarely has freezing temperatures or winter snowfall, it is distinguished, in part, by cold intolerant succulents (*Carnegiea gigantea*). In addition, the Sonoran Desert supports a variety of tree species (*Cercidium macrophyllum*, *Acacia greggii*, *Olneya tesota*, *Prosopis glandulosa*, and *Fouquieria splendens*), perennial grasses (*Pleuraphis rigida*), and shrubs (*Hyptis emoryi*, *Atriplex pycnantha*, and *Larrea tridentata*) (Nielson 1987; MacMahon 2000). The annual temperatures range from 12 °C to 30 °C (Smith and others 1997). While rainfall varies across the region, mean annual rainfall is 258-311 mm/year and occurs in a bimodal pattern (Bahre and Shelton 1993; MacMahon 2000). In contrast to the Mojave, summer rainfall in the Sonoran Desert is produced by the NAMS (Smith and others 1997). During the summer monsoon season (July-September), the Bermuda high pressure cell shifts from the Atlantic coast to the northwest. Meanwhile, land and ocean temperatures diverge, resulting in a shift in wind patterns. This brings in moisture mostly from the Gulf of Mexico and also from the Gulf of California giving way to convective storm patterns (Barry and Chorley 2003). Overall, this bimodal precipitation pattern leads to high plant diversity, including winter active as well as summer active species (Barbour and Billings 2000).

The Chihuahuan Desert is located at the wettest end of the precipitation gradient. This desert community exhibits higher elevations and cooler temperatures than the Mojave and Sonoran Deserts (Smith and others 1997). The majority of precipitation in this region falls during the summer monsoon season (July-September) as a result of the NAMS, but winter rainfall and snowfall results in a weak bimodal signal. Annual precipitation averages 250 mm/year (77-507 mm/year) and the mean annual temperature is 14.7 °C (13.5 °C-16.3 °C) (Wainwright 2006). These climate factors contribute to a marked increase in perennial grass dominance (*Bouteloua eriopoda*, *B. gracilis*, *Sporobolus* spp., *Pleuraphis jamesii*, *Muhlenbergia* spp., and *Aristida purpurea*) (MacMahon 2000; Whitford 2002; Peters and Gibbens 2006). Other dominant species are *Larrea tridentata*, *Gutierrezia sarothrae*, and *Ephedra trifurca*.

Throughout the southwestern United States, climate models consistently project increased aridity, seasonal shifts in precipitation, and more extreme precipitation events (Diffenbaugh and others 2005; Seager and others 2007; Schoof and others 2010). Already,



temperatures are increasing faster in the southwestern United States than anywhere in the nation (USGCRP 2009). While droughts and megadroughts are a trait of the regional climate, a poleward shift in the subtropical dry zones, for example, would give way to a drying pattern that the Southwest has never experienced (Fawcett and others 2011; Seager and others 2007). Moreover, with a novel base state of aridity, these ecosystems may not be able to recover from periodic drought events (Gutzler and Robbins 2010).

## Carbon Cycling in North American Warm Deserts \_\_\_\_\_

### ***Precipitation***

For each warm desert type, the majority of precipitation events are  $\leq 5$  mm and primarily increase microbial activity and  $\text{CO}_2$  efflux to the atmosphere, whereas precipitation events  $\geq 10$  mm are required for GPP to exceed  $R_{\text{eco}}$  (Huxman and others 2004; Loik and others 2004; Kurc and Small 2007; Svejcar and others 2008). Between rainfall events, GPP and microbial activity decrease although a substantial portion of deep autotrophic root respiration from  $>20$ -50 cm depth can occur in desert shrublands (Breecker and others 2012). At the soil surface, inorganic nitrogen, microbial biomass, and soil organic matter accumulate (Austin and others 2004). When dry soil is re-wet with a large ( $\geq 10$  mm) precipitation event, microbial activity quickly increases but net  $\text{CO}_2$  uptake is not immediate due to plant acclimation and growth of new roots and leaves (Ogle and Reynolds 2004). In the Sonoran and Chihuahuan Deserts, there is consistently a two- to three-day lag following a precipitation event before GPP surpasses  $R_{\text{eco}}$  losses (Potts and others 2006; Kurc and Small 2007; Scott and others 2010; Hamerlynck and others 2011). The delayed increase in NEE may also vary due to differences in soil moisture and the growing season. For example, Wohlfahrt and others (2008) observed a three- to four-day lag prior to an increase in NEE following a summer rainfall event in the Mojave Desert, while NEE increased within the same day following a spring rainfall event that occurred at peak soil moisture and primary productivity.

Regardless of plant community structure, similar to ANPP, the variability in NEE is mostly driven by large rainfall events (Kurc and Benton 2010). This trend is intuitive in the Mojave Desert where deeper rooted  $\text{C}_3$  shrubs are dominant, but it is less obvious in desert grasslands where the majority of  $\text{C}_4$  roots are located in the top 30 cm of the soil (Gibbens and Lenz 2001; McCulley and others 2004). Nevertheless, research indicates that a significant increase in  $A_{\text{net}}$  of the dominant Chihuahuan Desert grass, *Bouteloua eriopoda*, in response to small (5 mm) rainfall events does not always translate into a significant increase in ANPP or NEE (Kurc and Small 2007; Anderson-Teixeira and others 2011; Thomey and others 2011). Another factor that contributes to the variability in NEE is leaf area (Flanagan and others 2002; Polley and others 2010a). At the leaf level, research shows that native grass  $A_{\text{net}}$  development tracks leaf area in desert grasslands (Ignace and others 2007; Hamerlynck and others 2011). In a semiarid riparian ecosystem, the variability in GPP under drought conditions was attributed to leaf area in grassland as opposed to stomatal adjustments in woodlands (Jenerette and others 2009). Moreover, Comstock and Ehleringer (1986) deemed total plant canopy leaf area in *Encelia frutescens* as the main factor limiting whole-plant carbon gain. Likewise, at the community scale, Knapp and Smith (2001) also noted that ANPP is constrained by leaf area and plant density in Chihuahuan Desert grassland. Thus, the variability in NEE across the warm deserts is attributed to large precipitation events that extend periods of increased soil water availability leading to improved leaf area and net  $\text{CO}_2$  uptake. In contrast, small precipitation events can trigger a physiological response (e.g., increase  $A_{\text{net}}$ ) but are quickly lost by evapotranspiration. Although water is the most limiting resource in desert ecosystems, a threshold exists such that precipitation pulses may

initiate  $A_{\text{net}}$  and limited plant growth (ANPP) but net  $\text{CO}_2$  uptake does not surpass  $R_{\text{eco}}$  (Fernández 2007; Muldavin and others 2008; Kurc and Small 2007).

In addition to pulse size, antecedent soil moisture along with the seasonal timing of rainfall affects  $\text{CO}_2$  uptake. In the Sonoran Desert, the magnitude of the response of  $A_{\text{net}}$  was greater when water was added to dry soil when compared to wet soils where desert grasses had already reached their physiological capacity (Ignace and others 2007). Similarly, Patrick and others (2009) observed that the Chihuahuan Desert shrub, *Dasyliirion leiophyllum*, did not increase  $A_{\text{net}}$  when additional water was supplemented to wet soils. This trend is also reflected in NEE but for different reasons. For example,  $\text{CO}_2$  uptake was enhanced when a large rainfall event occurred following a pre-monsoon season drought because  $R_{\text{eco}}$  was limited likely by substrate supply or nitrogen availability (Austin and others 2004; Potts and others 2006). The response of  $\text{CO}_2$  uptake to the seasonal timing of rainfall is often confounded by the dominant plant type. For example, desert grassland ANPP is strongly correlated with summer (August) precipitation, and ANPP in desert shrublands responds mostly to winter precipitation (Cable 1975; Huenneke and others 2002; Muldavin and others 2008). This affects NEE in several ways. In desert grasslands, NEE will likely not increase if above-average rainfall is evenly distributed throughout the year when grasses are dormant rather than falling during the peak summer growing season (Mielnick and others 2005). On the other hand, desert shrublands act as a strong source of  $\text{CO}_2$  if below-average rainfall occurs during the winter season (Scott and others 2009). In general, soil moisture and seasonal patterns of precipitation affect the biotic response and drive the variability in NEE (Polley and others 2010b).

## $\text{CO}_2$

Arid-semiarid ecosystems are projected to be the most responsive to elevated  $\text{CO}_2$  concentrations largely because decreased  $g_s$  will lead to higher plant WUE (Mooney and others 1991; Melillo and others 1993; Smith and others 1997). In arid environments, these physiological adjustments have the potential to decrease ET and increase  $\theta$  (Morgan and others 2004). However, the effect of  $\text{CO}_2$  extends beyond WUE (Field and others 1995). For example, elevated  $\text{CO}_2$  concentrations are predicted to increase primary productivity, particularly in  $\text{C}_3$  species due to the suppression of oxygenase activity of Rubisco (Lambers and others 2008). Litter quality is another factor that may be impacted as higher  $\text{CO}_2$  concentrations increase C:N ratios and decrease decomposition as well as soil N availability (Strain and Bazzaz 1983; Mooney and others 1991). The impact of increased atmospheric  $\text{CO}_2$  concentrations will vary by ecosystem (Field and others 1995).

The most intensive experiment to study the response of arid ecosystems to elevated  $\text{CO}_2$  occurred in the Mojave Desert at the Nevada Desert Free-Air  $\text{CO}_2$  Enrichment Facility (NDFF). Experimental plots were continuously exposed to  $\text{CO}_2$  concentrations of  $\sim 550 \mu\text{mol mol}^{-1}$ . Following seven years of treatment, the highest mean daily NEE in elevated  $\text{CO}_2$  plots occurred during periods of peak plant cover or high soil moisture (Jasoni and others 2005). Soil moisture remained an important driver of the effects of  $\text{CO}_2$  throughout all studies at the NDFF. For instance, belowground root respiration of the dominant shrubs *Larrea tridentata* and *Ambrosia dumosa* was unaffected by elevated  $\text{CO}_2$  but, instead, reached peak rates when soils were near their maximum water content (Clark and others 2010). Although elevated  $\text{CO}_2$  slightly increased the C:N ratio of plant litter at the soil surface, decomposition was affected more by the proportion of non-decomposable (high lignin and secondary compounds) to decomposable species present in the litter, which varied with inter-annual differences in precipitation (Weatherly and others 2003). Additional data from the NDFF also indicate that  $\text{CO}_2$  exposure does not affect litter quality through increased C:N ratios (Billings and others 2003). On

the other hand, leaf-level photosynthesis was enhanced by elevated CO<sub>2</sub> (Hamerlynck and others 2000; Naumburg and others 2003; Houseman and others 2003). Moreover,  $g_s$  in *Larrea tridentata* decreased by 25-50% in response to elevated CO<sub>2</sub>, but these patterns were most pronounced ( $A_{net}$ ) in or limited (decreased  $g_s$ ) to years with above-average precipitation (Naumburg and others 2003). Similarly, elevated CO<sub>2</sub> increased aboveground production of *L. tridentata*, *Ambrosia dumosa*, and *Krameria erecta*, but this response was also limited to years with high rainfall (Smith and others 2000; Houseman and others 2006). Based on these results, an increase in atmospheric CO<sub>2</sub> concentrations in arid ecosystems may enhance carbon sequestration but only in years with above-average rainfall.

The degree to which CO<sub>2</sub> affects ecosystem processes depends on lower  $g_s$  and ET that combine to increase soil  $\theta$  (Field and others 1995). Overall at the NDFF, higher atmospheric CO<sub>2</sub> concentrations did not conserve soil  $\theta$  (Nowak and others 2004). However, soil  $\theta$  was higher under elevated CO<sub>2</sub> during wet years (Nowak and others 2004) and this corresponded with a decrease in  $g_s$  of the dominant shrub *Larrea tridentata* (Naumburg and others 2003). In addition, at the NDFF, down-regulation of photosynthesis occurred in *L. tridentata* and *Lycium andersonii* shrubs grown under elevated CO<sub>2</sub> concentrations (Huxman and others 1998; Pataki and others 2000; Hamerlynck and others 2002). This can occur in C<sub>3</sub> species when carbohydrates accumulate and directly inhibit photosynthesis or when less photosynthetic enzymes are produced (e.g., Rubisco) in response to elevated CO<sub>2</sub> concentrations (Stitt 1991). This response to long-term elevated CO<sub>2</sub> will decrease carbon sequestration capacity. Another factor to consider in a CO<sub>2</sub> enriched environment is that while lower  $g_s$  can lead to favorable WUE, the reduction in transpiration could increase leaf temperatures, offset reductions in  $g_s$ , and affect photosynthesis (Bazzaz 1990). Higher leaf temperatures did not occur for the species that exhibited decreased  $g_s$  at the NDFF (Nowak and others 2001). Generally, desert ecosystems may be exempt from this for the following reasons: (1) small leaves track air temperature due to high convective heat transfer, and (2) heatloss by transpiration in desert species is a minimal component in leaf energy balance due to water limitations (Lambers and others 2008). One last issue to consider is that increased primary productivity due to higher CO<sub>2</sub> can only be sustained with continued availability of soil N (Schaeffer and others 2007).

## Warming

In addition to precipitation and CO<sub>2</sub>, warmer temperatures can directly affect primary productivity and respiration or indirectly affect carbon balance by altering soil moisture, N availability, and species composition (Shaver and others 2000). In a meta-analysis from 85 warming studies established around the globe, Wu and others (2011) found that warming generally stimulated plant growth (above- and belowground net primary productivity), respiration, and NEE. Moreover, a 0.3-6.0 °C increase in temperature increased mean N mineralization by 46% across ecosystems ranging from tundra to grasslands (Rustad and others 2001). Unfortunately, desert ecosystems are rarely included in warming studies. However, we would expect that warmer temperatures alone would intensify soil water limitation and, therefore, decrease respiration and primary productivity, but these responses may be varied (Shaver and others 2000; Heimann and Reichstein 2008; Shen and others 2009). For example, in Chihuahuan Desert shrublands, van Gestel and others (2011) found that by reducing the daily range of soil temperature, soil moisture and soil respiration increased and N availability decreased while  $A_{net}$  of *Larrea tridentata* was not affected. Warmer minimum nighttime air temperatures (Figure 6) had an opposing effect on vegetation cover in Chihuahuan Desert grassland. A 2.5 °C increase in minimum nighttime air temperatures significantly increased cover





**Figure 6**—Wet and dry deposition collector (foreground) and a long-term climate change experiment (background) in mixed species composition desert grassland in the Sevilleta National Wildlife Refuge. The climate change experiment is designed to determine the individual and interactive effects of increased winter precipitation, nighttime warming, and nitrogen deposition on this desert grassland ecosystem. Variables measured in each replicate (N = 5) are: air and soil temperature, soil moisture, plant available soil nitrogen, aboveground net primary production (ANPP), and plant species composition. In addition, a subset of the treatment also continuously measures soil respiration. See <http://sev.lternet.edu/projects> for more information on this and other climate change related research being conducted as part of the National Science Foundation-funded Sevilleta Long-Term Ecological Research Program. Photo credit: Sevilleta LTER Program.

of the Chihuahuan Desert grass *Bouteloua eriopoda* and  $C_3$  shrub *Gutierrezia sarothrae* while warmer temperatures did not affect *Bouteloua gracilis* cover (Collins and others 2010). Also in the Chihuahuan Desert grassland, Bell and others (2008) found that while bacterial response tracked precipitation, soil temperature was the major determinant of fungal carbon use. Since community composition and microbial activity are sensitive to warmer temperatures in arid ecosystems, additional research is needed to fully understand how this element of climate change may re-shape these communities.

### **Environmental Hurdles**

Environmental hurdles limit our ability to increase carbon sequestration in arid-semiarid ecosystems. These barriers alter the carbon cycle, are often exacerbated by climate change, and will affect the options available to land managers to enhance carbon sequestration. In this section, we focus on invasive species and disturbance regime (periodic destruction or removal of ecosystem components [Hobbs 2009]) as it relates to grazing, fire, and the carbon cycle.

**Disturbance**—In many ecosystems, periodic disturbance is typically followed by a period of succession where an ecosystem is gradually restored to a similar pre-disturbance state (Hobbs 2009). Succession in arid-semiarid ecosystems, however, may not occur along this trajectory. Across the warm deserts, the capacity to regenerate from disturbance (e.g., resilience) is directly linked to resource availability (Brooks and Chambers 2011). As such, low primary productivity and annual rainfall in the Mojave Desert results in low resilience to disturbance, whereas greater primary productivity and annual rainfall leads to increased resilience to disturbance in the Chihuahuan Desert. Moreover, novel disturbances, such as increased fire frequency in the Mojave Desert, result in a succession process that is varied or unknown. For example, burned *Larrea tridentata* communities converged with unburned *L. tridentata* communities over 19 years, while burned and unburned *Coleogyne ramosissima* communities lacked convergence irrespective of time since fire (Engel and Abella 2011). In contrast, the debate continues over the role of grazing intensity and fire frequency in the Sonoran and Chihuahuan Deserts.

In general, fire and grazing can affect ecosystems through a variety of mechanisms that act on components of the carbon cycle. For example, both grazing and grazing exclusion have been found to promote shrub encroachment in Sonoran and Chihuahuan Desert grasslands (Asner and others 2004; Yanoff and Muldavin 2008; Browning and Archer 2011). The change in species composition from grassland to shrubland would be expected to affect spatial variability of soil organic carbon through changes in above- and belowground plant growth as well as N availability, especially if the shrub was a legume such as mesquite (Piñeiro and others 2010). Emerging evidence in the Chihuahuan Desert suggests that creosote bush shrublands provide a larger carbon sink than the black grama grasslands they are invading (Figure 7).



**Figure 7**—Shrub encroachment—invasion of native C4 grassland by native C3 shrub species—is occurring globally. In the Southwest, many grasslands are being invaded by creosote bush (*Larrea tridentata*, pictured) and mesquite (*Prosopis* spp.). Causes of shrub encroachment vary regionally but generally include interactions among several factors, such as overgrazing, drought, elevated atmospheric CO<sub>2</sub>, fire suppression, and global warming. At the Sevilleta National Wildlife Refuge, shrub encroachment into grasslands results in higher night time temperatures during winter, lower plant species diversity, increased soil resource heterogeneity, and increased organic matter and nutrient losses via runoff. However, evidence suggests that creosote dominated shrublands are a stronger carbon sink than the grasslands they replace. Photo credit: Sevilleta LTER Program.



Climate change is likely to increase fire frequency in the Southwest, which may facilitate a change in species composition by invasive species or be perpetuated by the presence of invasive grasses across the warm deserts (Abatzoglou and Kolden 2011). Although there exists positive feedback among climate, disturbance, invasive species, and the carbon cycle, changes in carbon cycling associated with disturbance are also significant in the absence of invasive species. A study by Vargas and others (2012) in Chihuahuan Desert grassland consisting largely of the native perennial grass *Bouteloua eriopoda* and others showed that fire decreased  $R_s$  and NPP by an average of ~70%. Another study found that the native Chihuahuan Desert grasses *B. eriopoda* and *Aristida purpurea* increased  $A_{net}$  and  $G_s$  in response to fire likely due to an increase in soil N availability (Allred and Snyder 2008). Additional studies are needed to fully understand how arid-semiarid ecosystems respond and recover from disturbance as climate change will likely increase the susceptibility of these ecosystems to disturbance and biological invasions.

**Invasive Species**—In this section, we define “invasive” as any native or non-native species that causes or is likely to cause social, economic, or ecological harm (Runyon and others 2012). Across the Mojave, Sonoran, and Chihuahuan Deserts, high temperatures, limited water availability, and nutrient deficient soils create an environment where a specific suite of native plants persist (Rao and others 2011). However, increasing  $CO_2$  concentrations, precipitation variability, and increasing N deposition modify the availability of limiting resources and may increase invasive species success in arid-semiarid ecosystems (Davis and others 2000). For example, when grown at elevated  $CO_2$  concentrations, plant density, biomass, and seed rain were significantly higher in the non-native annual *Bromus madritensis* ssp. *rubens* compared to native annuals during a wet year in the Mojave Desert (Smith and others 2000). The favorable response of *B. madritensis* ssp. *rubens* is attributed to a decrease in the energetic cost of aboveground biomass construction under elevated atmospheric  $CO_2$  concentrations (Nagel and others 2004). In another study, Suazo and others (2012) found that non-native annual species increased in density (*Schismus* spp.) and allocated more resources to seed production (*Brassica tournefortii*) in response to disturbance as well as supplemental watering. This indicates that invasive species establishment in desert ecosystems is also favored by multiple aspects of climate change.

Once established, invasive species can alter ecosystem processes that feed back to the climate system (Dukes and Mooney 1999). This has been observed with shrub encroachment in the Chihuahuan and Sonoran Deserts. When experimental rainfall is -80%, -50%, +50%, or +80% of ambient rainfall, the  $C_3$  shrub *Prosopis glandulosa* consistently maintains higher  $A_{net}$  and  $\Psi_{pd}$  compared with the  $C_4$  grass *Bouteloua eriopoda* in the Chihuahuan Desert (Throop and others 2012). Similarly, in the Sonoran Desert, *Prosopis velutina* sustained higher  $A_{net}$  and photosynthetic function across a greater temperature range and under dry pre-monsoon conditions compared with the native  $C_4$  grass *Sporobolus wrightii* (Barron-Gafford and others 2012). These results are especially interesting since  $C_4$  species usually maintain higher rates of  $A_{net}$  and WUE under warmer and drier environmental conditions (Lambers and others 2008). However, in the Barron-Gafford and others (2012) study, the deep-rooted  $C_3$  shrub outperformed the  $C_4$  grass because it had access to groundwater while significantly higher  $A_{net}$  in *S. wrightii* was limited to brief periods of higher water availability during the monsoon season. Experimental results also suggest that shrub encroachment can influence  $R_s$  because of shrub-induced changes in microsite conditions. Research by Cable and others (2012) showed that  $R_s$  was higher near the trunk and under the canopies of large *Prosopis velutina* but temperature sensitivity of  $R_s$  was greatest at the canopy edge. Furthermore,  $R_s$  under large *P. velutina* canopies was more similar to  $R_s$  under grass canopies as opposed to that measured under the canopy of medium



sized *P. velutina* (Cable and others 2012). The differences in Rs with varied microsite conditions are likely due to higher root biomass in large *P. velutina* or to differences in heterotrophic and autotrophic response patterns from the trunk to canopy edge. At the continental scale, shrub encroachment into Chihuahuan Desert grassland results in lower ANPP compared to mesic sites where shrubs have invaded (Huenneke and others 2002; Knapp and others 2008; Barger and others 2011). These differences in primary productivity and Rs in response to shrub encroachment stand to alter the source-sink strength of warm desert ecosystems.

## **Management and Policy Implications**

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### ***Carbon Sequestration and Land Management***

According to the studies, the elements of climate change will probably have contrasting effects on NEE and carbon uptake in the warm deserts. Nevertheless, increased precipitation variability will substantially affect the capacity of these water-limited ecosystems to exploit increased atmospheric CO<sub>2</sub> concentrations and persist during prolonged droughts. Certainly, climate change mitigation in arid-semiarid ecosystems has a low potential when compared with more productive ecosystems (Brown and others 2010). However, arid-semiarid systems cover ~45% of the terrestrial surface (Schimel 2010) and desertification is projected to increase (IPCC 2007), making management strategies that increase C sequestration or decrease carbon loss worth pursuing.

Among the elements of climate change, precipitation variability is the least predictable but remains the most influential component in arid-semiarid ecosystems. Consequently, the most beneficial and cost-effective land management strategies should work to restore degraded lands and improve their potential for storing atmospheric CO<sub>2</sub> (Follet and Reed 2010; Laca and others 2010; Piñeiro and others 2010), maximize carbon gain during peak productivity (Svejcar and others 2008), and combine eddy co-variance techniques with field based measurements and predictive modeling to understand how arid-semiarid ecosystems function and to predict effective management strategies (Brown and others 2010).

Presently, CO<sub>2</sub> concentrations are increasing by roughly 2 ppm annually while mean global temperatures are projected to increase 1.1-6.4 °C by the end of this century (IPCC 2007). Since CO<sub>2</sub> is the most prevalent greenhouse gas, prevailing scientific theory suggests reducing atmospheric CO<sub>2</sub> concentration via enhanced carbon storage may slow or reverse the rate of global climate change. As a result, some Federal land management agencies are now implementing management policies to increase carbon sequestration.

### ***The Climate Change Scorecard***

The USDA Forest Service developed the National Roadmap for Responding to Climate Change (<http://www.fs.fed.us/climatechange/pdf/Roadmapfinal.pdf>), which includes three interconnected responses:

1. Assess current risks, vulnerabilities, policies, and gaps in knowledge,
2. Engage employees and stakeholders to seek solutions, and
3. Manage for resilience, in ecosystems as well as in human communities, through adaptation, mitigation, and sustainable consumption strategies.

The Forest Service is using the Climate Change Scorecard to track progress on this Roadmap. Element Nine of the Climate Change Scorecard addresses carbon sequestration and provides fundamental questions for managers to address when reporting on

their accomplishments towards developing land management strategies in the context of climate change. The Scorecard defines the required baseline assessment of carbon stocks, including the consideration of management practices to increase carbon sequestration or reduce emissions. The scorecard evaluation process contains provisions for: (1) developing a baseline assessment of carbon stocks; (2) understanding how management practices and disturbances influence carbon stocks, sequestration, and emissions; and (3) balancing carbon stewardship with the management of other benefits. These provisions are linked to complicated biological processes, which are often poorly understood and represented by confounding research. However, helpful guidelines and generalizations can be made. The following section provides an overview of how management practices influence carbon sequestration. We focus on fire and grazing as management tools (Figures 7 and 8).

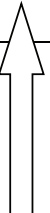

### ***Carbon Sequestration and Storage Defined***

Carbon sequestration includes any process for capturing and storing atmospheric CO<sub>2</sub> long term. Since a large portion of carbon resides in soils of arid-semiarid ecosystems, a clear definition and understanding of carbon sequestration in non-forest ecosystems is necessary to estimate the management effects on carbon stores (Table 1). Carbon pools contained in the soil emanate from both organic and inorganic sources. Soil carbon is most often characterized as SOC or SIC. SOC is a component of soil organic matter (SOM), which includes all materials of biological origin regardless of origin or state of decomposition (Baldock and Skjemstad 1999). Organic matter contains roughly 60% carbon (Bell and Lawrence 2009), meaning that soil with 1.7% SOM will yield roughly 1% SOC. Whereas SOC originates from carbon fixation during photosynthesis and microbial decomposition, SIC (e.g., CaCO<sub>3</sub>) is the result of mineral weathering.



**Figure 8**—Controlled burn in Chihuahuan Desert grassland at the Sevilleta National Wildlife Refuge, New Mexico. Photo credit: U.S. Fish and Wildlife Service.

Table 1. "Pools" (or types) of soil organic carbon (adapted from Liddicoat and Others 2010).

Organic carbon pool	Size	Stability	Turnover time	Key functions
<b>(1) Crop residues</b> Shoot and root residues on and in the soil	> 2mm	Labile (readily available)	Days	Provide energy and nutrients to biological processes; readily broken down providing soil conditions that favor soil biology.
<b>(2) Particulate organic matter (POM)</b> Smaller plant debris	0.05-2mm			These are broken down relatively quickly in suitable conditions but more slowly than crop residues. Important for soil structure, provision of energy for biological processes and nutrients.
<b>(3) Humus</b> Decomposed material dominated by molecules stuck to soil minerals	< 0.05mm			This plays a role in all key soil functions, but is particularly important in the retention and provision of nutrients (e.g. the majority of available N is found in the humus fraction).
<b>(4) Recalcitrant organic matter</b> Biologically stable, dominated by pieces of charcoal	Variable	Very stable / relatively inert	Hundreds of years	Decomposes very slowly and if present in large enough quantities can contribute to cation exchange capacity as well as controlling soil temperature.

SIC concentrations tend to be much more stable through time because inorganic carbon compounds are not available as microbial substrate. Given that most management techniques are administered to influence vegetation, it follows that the rate of organic carbon accumulation is directly influenced by management actions. As a result, this section focuses primarily on SOC, which is directly influenced by management activities.

Prior to understanding how management actions can alter carbon stores or developing a monitoring protocol, a baseline assessment is needed to estimate the current amount of carbon stored in an area. Since far more carbon is stored below- than aboveground (Allen-Diaz and others 1995), it follows that any baseline assessment should include a substantial focus on belowground carbon pools. Creating a baseline assessment of carbon stocks on rangelands poses unique challenges. Unlike forested landscapes, few agency standards or resources exist for collecting information on non-forest environments. For example, there exist no codified provisions for the FIA Program (FIA 2010) for evaluating non-forested stands. In addition, a greater proportion of carbon is stored belowground in rangelands compared with forests or woodlands, thereby requiring expensive soil analyses for quantifying baseline carbon stores. Methods for estimating carbon stores can be categorized as direct measures, indirect measures, and modeling. Each category includes advantages and disadvantages and varying levels of cost, complexity, and accuracy.

## Measuring Stored Carbon

Direct measures generally include field and laboratory measurements and eddy covariance techniques (Baldochi and others 1988; Verma 1990; Lee and others 2004; Burba and Anderson 2010). Laboratory measurements for soil carbon include techniques such as dry combustion or bulk density testing and are nicely described in Donovan (2012; measuring soil carbon change). Four promising field-based measurement techniques include Laser Induced Breakdown Spectroscopy (Cremers and others 2001), Inelastic Neutron Scattering (Wielopolski and others 2000), Mid-Infrared Reflectance Spectroscopy (McCarty and others 2002), and Visible-Near Infrared diffuse reflectance



spectroscopy. The USDA Natural Resources Conservation Service is systematically measuring soil carbon using the visible-near infrared method and it is conceivable that a similar approach could be used by Forest Service field units to fulfill the obligations of the Climate Change Scorecard.

Although most carbon resides belowground as SOC, some rangeland ecosystems exhibit a substantial amount of carbon stored aboveground. Aboveground carbon stores are not always considered in sequestration studies due to the relative ease with which shrubs can disappear from a stand. Nevertheless, many stands support significant aboveground carbon stores, which are important to evaluate when a full accounting of carbon stocks is required. Inventorying and monitoring aboveground carbon stocks can easily be achieved using destructive sampling or allometric equations (e.g., Means and others 1994; BIOPAK Software, <http://andrewsforest.oregonstate.edu/data/tools/software/biopak.cfm?topnav=149>) that relate stand structure to biomass components. Once a baseline assessment of carbon stores has been conducted, the more challenging task of monitoring could be considered.

### ***Monitoring Considerations***

Monitoring changes in carbon stores is critical to determining the efficacy of various types and levels of landscape treatments and disturbance regimes. Monitoring carbon stocks in rangeland ecosystems is inherently difficult because the change in mass of carbon is small relative to the current carbon store and usually takes years or decades before significant trends can be revealed.

SOC is often monitored using field or laboratory measures, indirect observations such as remote sensing, and ecosystem modeling (Brown and others 2010) or a combination of these techniques (Hunt and others 2004). Properly calibrated ecosystem simulation models, such as Comet-VR (<http://www.cometvr.colostate.edu/>), provide a suitable framework for estimating the effects of disturbances and land management techniques on carbon stores of rangelands. The most common rangeland management practices that have a meaningful impact on SOC are fire and herbivory. The effects of these practices on carbon stores and rates of sequestration are often inter-linked, confounded, and poorly understood across the wide range of non-forest ecosystems.

### ***Effects of Management Actions and Disturbances on Carbon Storage***

Fire in most non-forest systems has the obvious effect of reducing aboveground carbon stores (Figure 8). However, beyond the initial and obvious response, the effects of fire on carbon sequestration on non-forest landscapes are complicated to quantify. Fires can change plant diversity and structure by changing the availability of light, water, and nutrients. Such alterations in ecosystem structure and physical environment often modify primary productivity and nutrient fluxes that subsequently alter carbon storage in plants and soils (Ojima and others 1990; 1994; Rice and others 1998; Reich and others 2001). The direction and magnitude of these changes, however, depend on vegetation type, fire frequency, fire duration, and season of fire occurrence. Changes in SOC in response to fire are difficult to determine over short time periods due to large quantities of carbon stored in soil and because changes in SOC storage occur slowly relative to the quantity of SOC. In general, fires tend to enhance ANPP in grasslands and savannas during the first several growing seasons after the fire event (Rice and Owensby 2000; Ansley and others 2002; Santos and others 2003; Bremer and Ham 2010). Fires also appear to accelerate root production, increase root biomass, and increase C:N ratios of root tissue (Blair 1997; Rice and others 1998; Johnson and Matchett 2001; Reich and

others 2001) in some grasslands. While increased production may be observed over short time periods, it does not seem to cause commensurate increases in SOC over longer time periods (up to 50 years) (Sharrow and Wright 1977; Ansley and others 2010). Indeed, most studies indicate that C stocks only slightly decrease or demonstrate no change for up to 50 years of annual-biennial fire treatment (Rice and Owensby 2000). Despite this generality, characterizing these effects in a specific manner is complicated by a lack of controlled studies of sufficient duration for observing changes in SOC in response to fire characteristics. What's more, most studies aimed at fire effects on carbon dynamics have been conducted in relatively productive systems such as tallgrass prairie, which shows little decline in SOC with repeated fires (Rice and Owensby 2000).

These generalities depend on many other factors, resulting in complicated feedbacks and interactions between fire and other disturbance types. A lack of grazing can alter species composition and has been postulated to enhance shrub encroachment on semi-desert and desert grasslands (Asner and others 2004; Yanoff and Muldavin 2008; Browning and Archer 2011). Altered species composition modifies fire frequencies and intensities, which affect SOC through changes in above- and belowground primary production. Similarly, the ultimate effects of herbivory are not well understood and vary between and even within various rangeland ecosystems, making generalities or widely applicable guidelines difficult to develop.

One overarching factor, however, is that rangelands function as carbon sinks for relatively short periods (Svejcar and others 2008). This fact requires managers to thoughtfully consider management practices that do not impede sequestration during critical times. Sustaining carbon uptake relies on growth and development of vegetation, which requires careful balance with grazing, which affects C sequestration (Derner and Schuman 2007). In both short- and mixed-grass, grazing at light, moderate, and heavy stocking rates tends to increase SOC (Derner and others 1997, 2006; Reeder and Schuman 2002; Reeder and others 2004) in the surface 30 cm, while moderate and heavy grazing over many decades appears to increase SOC down to the 60-cm level in northern mixed-prairie (Liebig and others 2010). Although grazing can increase SOC, annual production can decrease as a result of altered species composition (Figure 9).



**Figure 9**—A lone steer grazing in Chihuahuan Desert on the Jornada Experimental Range in southern New Mexico. Photo credit: Scott Collins.

For example, heavy stocking rates can increase the relative proportion of warm season species that translocate greater amounts of C to belowground structures, often resulting in increased SOC (Coupland and Van Dyne 1979; Schuman and others 1999; Derner and Hart 2007). The general increase in SOC observed in the previously mentioned studies, however, must be interpreted with caution and will not always be applicable. For example, when heavy grazing is coupled with sustained or severe drought, significant losses of SOC can occur (Ingram and others 2008). This supposition is strengthened by Zhang and others (2010) who reported that more xeric rangelands tend to be in C balance or are C sources whereas more mesic rangelands function primarily as C sinks.

### ***Balancing Carbon Sequestration with Other Services***

Carbon sequestration and storage, as a greenhouse gas mitigation strategy, can be viewed as an ecosystem service. Management actions influencing carbon storage and emissions (i.e., encouraging shrub encroachment into grasslands) must ultimately be balanced with the need to sustainably produce other goods and services such as wildlife habitat, livestock, clean air and water, aesthetic properties, and other natural amenities (Figure 9). Addressing this need requires both the art and science of range management in addition to more comprehensive research aimed at understanding tradeoffs between carbon storage and other products or services. In many cases, production of multiple goods and services can be simultaneously enhanced through proper management techniques. For example, nutrient cycling (Rau and others 2010), forage (Sandell and Knezevic 2010), biotic integrity, biodiversity (Catovsky and others 2002), and carbon sequestration (Rau and others 2010) can be enhanced or maintained through removal of some invasive species. In contrast to this type of win-win scenario, tradeoffs between various management outcomes will depend on resource values and sometimes produce one-sided outcomes.

As an example, in the Chihuahuan Desert, consider the tradeoff between increased carbon capture and storage versus water abundance and availability. In some regions, simply aiding the spread and rate of encroachment by shrubs such as *Juniperus* or *Prosopis* species can capture more carbon. However, shrub encroachment leads to lower biodiversity, higher regional temperatures, increased surface runoff and erosion, and lowered water tables that could ultimately lower the amount of available surface water.

The relative value of various goods and services in the future will likely change with our changing climate, but the magnitude of these changes is subject to public perceptions as well as environmental, political, and socio-economic factors. Thus, it is conceivable that future carbon capture and storage may be less of a priority compared to a dependable supply of clean water or open space and winter range for ungulates (Figure 10).

Balancing societal needs and desires with ecological constraints is not easy, and the added consideration of carbon storage creates a multi-dimensional problem for which decision support systems may be invoked to understand tradeoffs between potential management actions. Toward this goal, the Sustainable Rangeland Roundtable, a group of private and public organizations and agencies, developed a forum to discuss rangeland sustainability and assessment to integrate social, economic, and ecological disciplines related to rangelands. In this regard, the Integrated Social, Economic, and Ecologic Conceptual Framework created by the Sustainable Rangeland Roundtable enables assessment of tradeoffs and implications of management decisions in a multi-dimensional manner. The likelihood of the Forest Service or Bureau of Land Management achieving carbon storage planning goals will be increased by use of decision support systems capable of integrating simultaneous multi-dimensional analyses.





**Figure 10**—A male mule deer (*Odocoileus hemionus*) in desert grassland co-dominated by *Bouteloua gracilis*, *B. eriopolda*, and *Sporobolus* spp. at the Sevilleta National Wildlife Refuge, New Mexico. Photo credit: Sevilleta LTER Program.

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### ***Table of Abbreviations***

$\theta$	Soil water content
$A_{\text{net}}$	Photosynthesis
ANPP	Aboveground net primary productivity
C	Carbon
$\text{CaCO}_3$	Calcium carbonate
$\text{CO}_2$	Carbon dioxide
ET	Evapotranspiration
FIA	Forest Inventory and Analysis
GPP	Gross primary productivity
$G_s$	Leaf conductance
mm	Millimeters
N	Nitrogen
NAMS	North American Monsoon System
NDFE	Nevada Desert Free-Air $\text{CO}_2$ Enrichment Facility
NEE	Net ecosystem exchange
Pg	Petagram
$R_{\text{eco}}$	Ecosystem respiration
$R_s$	Soil respiration
SIC	Soil inorganic carbon
SOC	Soil organic carbon
SST	Sea surface temperature
WUE	Water use efficiency







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