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Greenhouse Gas Mitigation Opportunities in California Agriculture

Review of California Rangeland Emissions and Mitigation Potential

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ABSTRACT

Rangelands cover approximately 50% of California and have considerable potential to mitigate climate change. Several management strategies offer opportunities to build soil carbon and reduce greenhouse gas emissions. Grazing management can increase soil carbon, but significant uncertainties remain and best management practices are unknown. Long-term, well-replicated studies are urgently needed to explore the potential of grazing management for climate change mitigation. Organic amendments, particularly compost, can enhance biomass and sequester carbon on grasslands while reducing emissions from the waste sector. This strategy shows significant potential but requires additional research, particularly in arid rangelands. High-efficiency synthetic fertilizer use, plant community management, fire management, and irrigation can also influence soil carbon; however, these strategies could be challenging to scale up over large areas, and their net greenhouse gas impacts are uncertain. Remote sensing, biogeochemical modeling, and life-cycle assessments should be leveraged to identify and implement mitigation strategies.

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CONTENTS

INTRODUCTION AND OBJECTIVES	5
RANGELAND DEFINITIONS, EXTENT, AND SIGNIFICANCE	6
RANGELAND SOIL CARBON AND GHG FLUXES: OVERVIEW.....	7
<i>Soil Carbon Pools, Soil CO₂ Fluxes, and Net Ecosystem Exchange</i>	<i>8</i>
Net Ecosystem Exchange	9
Soil Carbon Fluxes and Possible Drivers	10
<i>Summary of Key Findings: Soil C and CO₂ Dynamics</i>	<i>11</i>
<i>Methane and Nitrous Oxide Fluxes from Rangelands</i>	<i>12</i>
<i>Summary of Key Findings: Soil CH₄ and N₂O Dynamics</i>	<i>12</i>
RANGELAND MANAGEMENT: STRATEGIES AND IMPACTS.....	13
<i>Grazing.....</i>	<i>13</i>
<i>Fire.....</i>	<i>14</i>
<i>Soil Amendments.....</i>	<i>16</i>
<i>Cultivation.....</i>	<i>17</i>
Mowing.....	17
Aeration and Tillage.....	18
Irrigation.....	18
<i>Plant Community Composition</i>	<i>19</i>
<i>Summary of Key Findings: Rangeland Management</i>	<i>20</i>
TOOLS FOR MANAGEMENT DECISIONS: MODELING AND REMOTE SENSING.....	21
<i>Ecosystem and Biogeochemical Models</i>	<i>21</i>
<i>Life-Cycle Assessments</i>	<i>22</i>
<i>Remote Sensing.....</i>	<i>23</i>
KEY FINDINGS AND RESEARCH PRIORITIES	23
<i>Key Findings</i>	<i>23</i>
<i>Research Priorities</i>	<i>24</i>
REFERENCES.....	26

INTRODUCTION AND OBJECTIVES

Rangelands cover more than 22 million hectares (ha) in California, or 40–50% of the state’s land area (Brown et al. 2004). Globally, rangelands are critical to the carbon (C) cycle (Ogle, Conant, and Paustian 2004), storing about one-third of the terrestrial soil C pool (Jobbagy and Jackson 2000) over an area of approximately 3.3 billion ha. The large extent of rangelands, coupled with their propensity to store C in soils, suggests considerable C sequestration potential and thus opportunities for climate change mitigation. Numerous ecosystem services can be protected or enhanced by conserving or improving rangeland soils (Havstad et al. 2007). Management practices that conserve and enhance C storage in rangelands as well as prevent losses to the atmosphere can help mitigate climate change while enhancing sustainability under future climate scenarios.

Carbon cycling and greenhouse gas (GHG) fluxes are highly sensitive to climate in California’s rangelands (Ma, Baldocchi, Xu, and Hehn 2007; Jackson et al. 2007; Chou et al. 2008; Schwalm et al. 2012; Grant, Baldocchi, and Ma 2012). In the western United States, climate change is likely to occur even if substantial reductions to GHG emissions are achieved (Solomon et al. 2007). Air temperatures in the southwestern United States are predicted to rise 1.5–4.5 °C by 2099 (Cayan et al. 2008). Impacts on annual precipitation and precipitation patterns remain uncertain, although recent work suggests that a decline in mean annual rainfall is likely (Seager and Vecchi 2010). In general, variability in precipitation is predicted to increase, as is the frequency and intensity of extreme precipitation events (Solomon et al. 2007; Kerr 2008). Such changes in climate are expected to have particularly strong impacts on arid and semi-arid ecosystems of the western United States (Archer and Predick 2008; Backlund, Janetos, and Schimel 2008), with implications for water regimes, fire risk, soil C storage, and GHG emissions. Grasslands and shrublands are likely to expand and may experience shifts in plant community composition due to climate change (Pan et al. 1998), which, in turn, are likely to feed back directly and indirectly on C and GHG dynamics (Zavaleta and Kettley 2006). Climate change, coupled with land use patterns, is creating novel ecosystems that may require new and innovative management approaches.

Although rangelands store a regionally important pool of C, they are a relatively small contributor to California’s GHG emissions; the greatest emissions associated with rangelands likely come from livestock enteric fermentation and manure management (Pitesky, Stackhouse, and Mitloehner 2009). These emissions are included in inventories as part of the agricultural sector, which contributes an estimated 7% of GHG emissions in California (32.4 million t CO₂-eq¹; CARB 2013). The emissions from livestock enteric fermentation and manure management make up 60% of California’s agricultural emissions (CARB 2013); although not directly from rangelands, these livestock-related emissions could perhaps be mitigated by rangeland management. The primary focus of this report, however, is C storage and GHG dynamics on rangelands.

Understanding the C and GHG dynamics on rangelands under current and future conditions is necessary to identify gaps in knowledge and opportunities for management-based climate change mitigation. The objective of this report is to synthesize what is known about GHG mitigation through the management of rangeland systems in California. California’s Mediterranean climate, with wet winters and extended summer droughts, differentiates many of its rangelands from those in other climates. Therefore, this study

¹ The CO₂-equivalents unit (CO₂-eq) is used for reported values that consider the combined global warming potential (GWP) from CO₂, methane (CH₄), and nitrous oxide (N₂O). Carbon dioxide has a GWP of 1 (1 g CO₂-eq = 1 g CO₂). To convert C from CO₂ into CO₂-eq, multiply by 3.67 (44.01 g CO₂/12.01 g C). To convert C from CH₄ into CO₂-eq, multiply by 1.33 (16.04 g CH₄/12.01 g C in) and then by the GWP (1 g CH₄ = 25 g CO₂-eq on a 100 y timeframe; Solomon et al. 2007). To convert N from N₂O into CO₂-eq, multiply by 1.57 (44.01 N₂O/28.01 g N) and then by the GWP (1 g N₂O = 298 g CO₂-eq on a 100-year timeframe; Solomon et al. 2007). Note that 1 Mg = 1,000,000 g = 1 t.

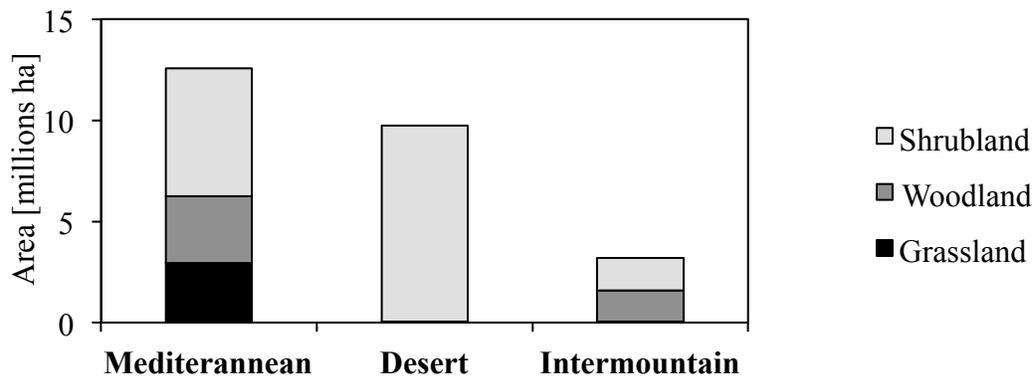
reviews the body of research on global rangelands but focuses on California and studies from regions with similar climate regimes. It concludes with a summary of key gaps in knowledge and future opportunities for climate change mitigation in rangeland systems.

RANGELAND DEFINITIONS, EXTENT, AND SIGNIFICANCE

This study defines *rangeland* broadly as land on which plant cover (climax, sub-climax, or potential) is composed principally of grasses, grass-like plants, forbs, or shrubs suitable for grazing and browsing, including native and introduced plant species (USDA 2009). *Grassland* is rangeland where the plant cover is primarily grasses and forbs. *Pastureland* is land managed primarily for the production of non-native forage plants, consisting of a single species, grass mixture, or grass-legume mixture; pasturelands are often distinguished from rangelands by the intensity of management, particularly chemical applications (USDA 2009). In this study, California rangeland includes grasslands, oak woodlands, chaparral, and some forested areas, wetlands, and deserts with the potential to be used for grazing. The drained wetlands and peatland pastures of the Sacramento–San Joaquin Delta (originally a 140,000 ha tidal marsh) also contain land managed as rangelands.

California’s diverse rangelands include many different plant communities and climate conditions, which impact soil C stocks and GHG fluxes. Thus, different rangeland types are likely to respond differently to management approaches. Estimates of total California rangeland vary among classification systems (Brown et al. 2004). An estimate of 25.5 million ha includes Mediterranean, desert, and intermountain/high-elevation cover types (Figure 1) (UCSB 1998; Mayer and Laudenslayer 1988; UCD 2011; Brown et al. 2004). This figure includes 11.9 million ha of Mediterranean rangelands (23% grasslands, 27% woodlands, and 50% shrublands, including chaparral and coastal scrub), 9.8 million ha of desert shrublands (primarily desert scrub), and 3.2 million ha of intermountain rangelands (49% dominated by juniper, 50% sagebrush, and other shrublands). A smaller estimate of rangeland extent (16.7 million ha) was calculated by the 2007 USDA ERS Major Land Uses survey and included only grassland pasture and range (11.1 million ha, including shrublands), cropland used as pasture, and grazed forest land (5.2 million ha of oak woodland and other wooded ecosystems with a grass understory; Nickerson, Ebel, Borchers, and Carriazo 2011).

Figure 1. Spatial Extent of Mediterranean, Desert, and Intermountain Rangelands in California and the Dominant Vegetation Type



Sources: Brown et al. (2004), UCD (2011), Mayer and Laudenslayer (1998).

Note: Grasslands are predominantly annual grasses; woodlands primarily consist of oaks with an annual grass understory. Rangelands with different cover types have different soil C stocks and GHG fluxes and are likely to respond differentially to management approaches.

In California, non-native annual grasslands invaded significant regions of the coast and Central Valley in the 1800s and replaced both native perennial-dominated grasslands (Biswell 1956; D’Antonio et al. 2007) and shrublands (Wolkovich et al. 2010; Engelberg et al. 2013). This invasion has been extensive; less than 2% of California’s Mediterranean grasslands are currently dominated by native perennial grasses (UCD 2011). Perennial grasses typically have comparatively deeper root systems, denser aboveground biomass, and higher root production—traits that enable perennials to survive the summer drought as well as to potentially increase soil C (Koteen, Baldocchi, and Harte 2011), although not in all cases (Wolkovich et al. 2010). Annual grass invasions in California’s shrublands have been linked to many types of disturbances (Eliason and Allen 1997; Fleming, Diffendorfer, and Zedler 2009). Natural disturbances including fire and grazing often lead to succession by native species, whereas mechanical disturbances such as disking, plowing, or cultivation tend to promote the establishment of non-native species (Engelberg et al. 2013). In California coastal scrub, annual grass invasions have resulted in increased soil C storage (Wolkovich et al. 2010); this effect differs from the effects of grass invasions in the Great Basin, possibly due to different fire regimes (Bradley, Houghton, Mustard, and Hamburg 2006; Prater, Obrist, Arnone, and DeLucia 2006).

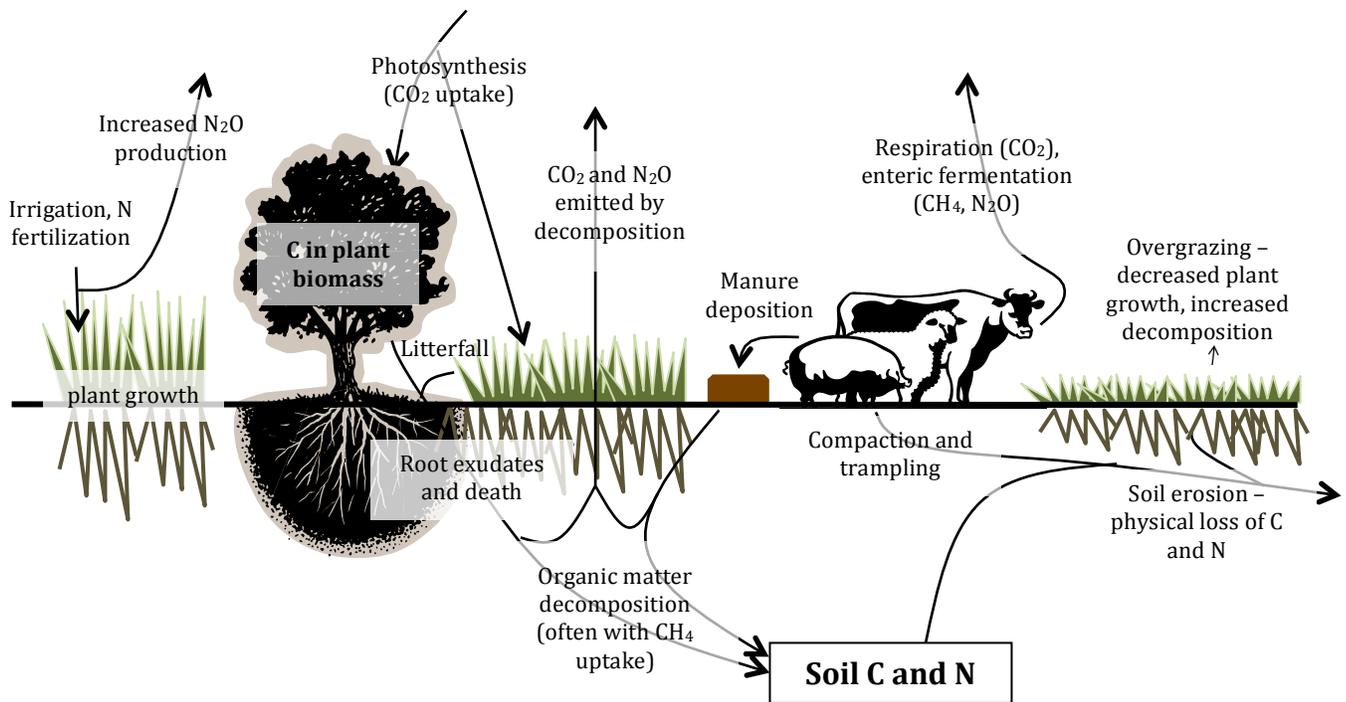
Woody encroachment into grasslands, documented in both northern and southern California (DeSimone and Zedler 2001), is part of a global phenomenon that has been attributed to changes in air temperature, precipitation patterns, elevated CO₂, and management (grazing, fire suppression, herbivore elimination) (Archer, Schimel, and Holland 1995). In California, the mechanisms leading to coastal scrub expansion are dependent on local conditions, including above-average rainfall, gopher activity, grazing, and lack of fire (DeSimone and Zedler 2001). Woody expansion into grasslands tends to increase aboveground C pools but has variable impacts on belowground C (Knapp et al. 2008; Asner et al. 2004), decreasing soil C storage in some ecosystems (Jackson et al. 2002).

Oak woodlands are of both ecological and cultural importance to California due to the presence of several endemic oaks, *Quercus douglasii* and *Quercus lobata* Ne’e (blue and valley oak, respectively). Oak woodlands experience prolonged summer drought annually, which they mitigate through groundwater uptake (Miller et al. 2010). Compared with open grasslands, California oak woodlands have lower albedos, lower radiative surface temperatures, higher net radiation, greater sensible heat fluxes, and greater evaporation, all of which impact vegetation and productivity (Baldocchi, Xu, and Kiang 2004). Oak woodlands have been threatened by clearing, sudden oak death, and climate change (Kueppers et al. 2005). Removal of oaks impacts ecosystem C storage, because soil pools of both C and nitrogen (N), a commonly limiting nutrient, tend to be greater under oak canopies than in adjacent grassland (Dahlgren, Singer, and Huang 1997).

RANGELAND SOIL CARBON AND GHG FLUXES: OVERVIEW

Rangeland soils store significant C in soil organic matter and act as both a source and a sink of greenhouse gases (Figure 2). Soil C and GHG fluxes vary widely in space and time, driven by a combination of climate, soil characteristics, and management practices (Silver, Ryals, and Eviner 2010; Conant, Paustian, and Elliott 2001; Soussana, Tallec, and Blanfort 2010; Paustian, Collins, and Paul 1997). Research documenting soil C pools, net ecosystem exchange, and GHG fluxes of rangeland ecosystems—especially that focusing on California—is overviewed here. This review is intended to highlight what is known about the potential for rangelands to store C and to mitigate GHG fluxes under current and changing climate and environmental conditions (e.g., rainfall, temperature, CO₂ fertilization, N deposition). Such information is required to identify mitigation opportunities on available landscapes, to identify knowledge gaps, and to inform research priorities. The specific impacts of management practices that can alter soil C and GHG fluxes are reviewed in the following section.

Figure 2. Schematic of the Major Pools of Carbon and Nitrogen Showing Major Fluxes of Greenhouse Gases Associated with California's Rangelands



Note: Flux arrows are not quantitative. Typically the largest CH_4 and N_2O fluxes in rangelands are derived from enteric fermentation. Soils are the largest C and N pool in rangelands, and many have considerable additional C storage potential.

Soil Carbon Pools, Soil CO_2 Fluxes, and Net Ecosystem Exchange

Rangelands store approximately 30% of the global soil organic C pool (Schuman, Janzen, and Herrick 2002; Follett and Reed 2010; Scurlock and Hall 1998; Jobbagy and Jackson 2000). Soil texture, mineralogy, and nutrient content as well as plant community composition, climate, and management are often used to predict patterns in soil C storage at a range of spatial and temporal scales (Parton, Schimel, Cole, and Ojima 1987; Burke et al. 1989; Conant, Paustian, and Elliott 2001; Silver, Ryals, and Eviner 2010). Carbon is acquired directly by rangeland plants through assimilation of CO_2 via photosynthesis and indirectly via livestock manure deposition. Carbon can be lost through plant and microbial respiration, erosion, burning, leaching, harvesting, and grazing. The magnitude and direction of CO_2 effluxes over annual and interannual timeframes are affected by vegetation type, climate (Hunt et al. 2004; Ciais et al. 2005; Gilmanov et al. 2007; Soussana et al. 2007), and management (reviewed by Soussana, Tallec, and Blanfort 2010). Carbon sequestration occurs when net C gains exceed losses over a specific timeframe.

Relatively few studies have reported measurements of soil C in rangeland ecosystems. In a meta-analysis of soil C data from California's annual grasslands and oak woodlands, 15 studies with a total of 48 unique soil profiles were identified (Silver, Ryals, and Eviner 2010). Soil C content (0–50 cm depth) spanned 28 to 300 t C ha^{-1} , with a mean of 90 t C ha^{-1} (excluding one outlier). Soil texture can be a particularly strong predictor of soil C pools (Schimel et al. 1994); however, soil C storage was only weakly correlated with clay content (Silver, Ryals, and Eviner 2010). Thus, differences in management practices were suggested as the likely driver for the observed range (Silver, Ryals, and Eviner 2010). For perennial grasses and

shrubs, the age of the vegetation may also affect C storage. For example, soil C concentration declined significantly with shrub stand age in subsurface (30–40 cm) chaparral soils; two two- to three-year-old stands had mean soil C concentrations of about 0.7%, whereas two older stands had 0.3–0.5% C (Vourlitis, Zorba, Pasquini, and Mustard 2007). In a regional sampling survey of grasslands in Marin and Sonoma counties, soil C pools ranged from 100 to 300 t C ha⁻¹ in the top meter of soil (Silver et al. in prep). On pasture-based dairies in Marin and Sonoma counties, Owen and Silver (in prep) measured higher soil C content ranging from 60 to 222 t C ha⁻¹ in the top 50 cm. The grazed drained peatlands in the Bay Delta contain significantly more C than other California rangelands, with recent measurements of approximately 486 t C ha⁻¹ in the top 60 cm (Hatala, Detto, and Baldocchi 2012). However, these soils have been rapidly losing C since the 19th century (see below; Hatala, Detto, and Baldocchi 2012; Deverall and Leighton 2010). Relative to other rangeland C stocks, mean California rangeland C stocks (50 t C ha⁻¹ in the top 20 cm; Silver, Ryals, and Eviner 2010) fell toward the middle of the range measured from 500 sites in the Great Plains (10–90 t C ha⁻¹ in the top 20 cm; Burke et al. 1989). Few studies have compared arid and semi-arid systems, but one such study in Spain found that arid rangelands had significantly less soil C than nearby semi-arid rangelands (Ruiz Sinoga, Pariente, Diaz, and Martinez Murillo 2012).

In addition to organic C, many arid and semi-arid soils in southern California are known to contain significant amounts of inorganic C in carbonates (Machette 1985; Monger and Martinez-Rios 2001). Carbonates accumulate in semi-arid and arid soils as a byproduct of weathering under very dry conditions (Monger and Martinez-Rios 2001), and these soils can also retain relatively more inorganic C from parent material (Machette 1985; Trueman et al. 2009). Inorganic C can be lost through wind and water erosion, a process accelerated with surface exposure (Emmerich 2003; Serna-Perez, Monger, Herrick, and Murray 2006; Tamir et al. 2011). Management can influence inorganic C stocks (Monger and Martinez-Rios 2001) by affecting the concentrations of Ca or HCO₃⁻ in soil moisture, the precursors to carbonate formation. A study on shortgrass steppe in Colorado revealed that a heavily grazed site had significantly more inorganic C than a nongrazed site (by 16.3 t C ha⁻¹ within the top 90 cm; Reeder, Schuman, Morgan, and LeCain 2004). More research is needed to quantify the amounts of C stored in inorganic pools in California soils.

Net Ecosystem Exchange

Gains and losses of C from rangeland ecosystems can be quantified by measuring the net CO₂ flux across the ecosystem-atmosphere boundary, commonly termed the *net ecosystem exchange* (NEE). Net ecosystem exchange is the difference between gross primary production and ecosystem respiration (including autotrophic and heterotrophic respiration). It can be measured with eddy flux techniques that make continuous measurements over large areas (Baldocchi 2008). The annual grasslands and woodlands of California are underrepresented by eddy flux networks (Baldocchi 2008). However, flux towers have been installed over other rangeland systems as part of the AmeriFlux network (<http://ameriflux.lbl.gov>) and international efforts. The MexFlux network includes towers over semi-arid shrublands and grasslands, oak woodland, and arid rangeland (Vargas and Yopez 2011).

Studies of NEE in California include research conducted over an oak savanna and neighboring grassland in the north central region (Xu and Baldocchi 2003, 2004; Xu, Baldocchi, and Tang 2004; Baldocchi, Tang, and Xu 2006; Ma, Baldocchi, Xu, and Hehn 2007; Ma et al. 2012), research along an elevation gradient in the southeastern region (Goulden et al. 2012; Fellows and Goulden 2013), measurements over a southern California chaparral ecosystem (Claudio et al. 2006; Luo et al. 2007), and research over a peatland pasture in the Sacramento-San Joaquin Delta (Nieveen, Campbell, Schipper, and Blair 2005; Rogiers et al. 2008; Teh et al. 2011; Hatala, Detto, and Baldocchi 2012). With the exception of the peatland pasture, rangelands were CO₂ sources and sinks of similar magnitude (by approximately ±7 t CO₂-eq ha⁻¹ y⁻¹, depending on the year). Studies generally have not accounted for differences in soil characteristics or management, both of which can significantly impact C storage and loss.

Net ecosystem exchange in California rangelands, and in rangelands globally, is highly sensitive to climate variability both within and across years, and particularly sensitive to patterns in rainfall seasonality (Xu and Baldocchi 2004; Soussana et al. 2007; Gilmanov et al. 2010; Ma, Baldocchi, Xu, and Hehn 2007; Ma et al. 2012). During a once-per-century drought year (2002), the strength of the C sink of a southern California chaparral decreased significantly to $-0.7 \text{ t CO}_2\text{-eq ha}^{-1} \text{ y}^{-1}$ from a five-year range of -3.7 to $-6.2 \text{ t CO}_2\text{-eq ha}^{-1} \text{ y}^{-1}$, and the site became a large C source in the following year, emitting $7.3 \text{ t CO}_2\text{-eq ha}^{-1} \text{ y}^{-1}$. Grasslands tend to be less resistant to drought than shrublands and woodlands due to shallower rooting systems (Xu and Baldocchi 2003; Ma, Baldocchi, Xu, and Hehn 2007). Oak savanna was a net C sink ranging from -2.2 to $-5.9 \text{ t CO}_2\text{-eq ha}^{-1} \text{ y}^{-1}$ over a six-year study period, while a neighboring grassland ranged from a net sink of $-3.3 \text{ t CO}_2\text{-eq ha}^{-1} \text{ y}^{-1}$ to a net source of $5.1 \text{ t CO}_2\text{-eq ha}^{-1} \text{ y}^{-1}$ (Ma, Baldocchi, Xu, and Hehn 2007). Isolated summer rainstorms led to pulses in soil respiration, increasing the source term (Xu, Baldocchi, and Tang 2004; Baldocchi, Tang, and Xu 2006; Chou et al. 2008). In a rainfall manipulation experiment, water added early and late in the water year (i.e., September and May–July) increased soil C losses from microbial respiration and associated decomposition of C stored in soils (Chou et al. 2008), whereas increased rainfall during the rainy season had little effect on C pools and fluxes.

Globally, rangelands tend to act as C sinks but become sources following major disturbances such as overgrazing, mowing, fire, or extensive drought (Gilmanov et al. 2007; Zhang et al. 2010; Svejcar et al. 2008). For example, European grasslands went from a net sink of $-23.5 \text{ CO}_2\text{-eq ha}^{-1} \text{ y}^{-1}$ to a net source of $5.9 \text{ CO}_2\text{-eq ha}^{-1} \text{ y}^{-1}$ with drought events; organic-rich soils also tended to experience net emissions (Gilmanov et al. 2007). Compared with rangelands in the eastern United States, western rangelands appeared to be particularly sensitive to drought (Zhang et al. 2011), potentially due to their drier background conditions. Drought can lead to widespread tree mortality and changes in plant community composition, both of which can affect C emissions (Claudio et al. 2006; Fellows and Goulden 2013). Photodegradation of litter following the resumption of rain after long dry periods can be an important source of ecosystem respiration (Ma et al. 2012). Temperature is also an important driver of ecosystem C fluxes, because both gross primary production and ecosystem respiration are sensitive to temperature (Yuan et al. 2011). Other potential predictors of rangeland C fluxes include temperatures before oak leaf-out (cool temperatures delay leaf-out and shorten the growing season), cloudiness (clouds increase diffuse light and enable photosynthesis, but lower air temperatures lower tree respiration), and above-average growth in the preceding season (large amounts of leaf litter can limit new grass growth) (Ma, Baldocchi, Xu, and Hehn 2007). The impacts of increased droughts, heat waves, or other disturbances may depend more on timing than magnitude (Craine et al. 2012, 2013).

Drained wetland soils rich in organic matter can be large net CO_2 sources (Nieveen, Campbell, Schipper, and Blair 2005; Rogiers et al. 2008), including soils from the Sacramento-San Joaquin Delta (6.6 to $11.0 \text{ t CO}_2\text{-eq ha}^{-1} \text{ y}^{-1}$; Hatala, Detto, and Baldocchi 2012). The delta has lost approximately 136 million t C (499 million t $\text{CO}_2\text{-eq}$) since it was drained in the late 19th century (Crooks 2009) with rates of approximately 2400 t C ha^{-1} between 1926 and 2006 (Deverall and Leighton 2010).

Soil Carbon Fluxes and Possible Drivers

Many rangelands in California (Chou et al. 2008; Koteen, Baldocchi, and Harte 2011) and globally (Sanderman and Baldock 2010; Bridges and Oldeman 1999; Bai, Dent, Olsson, and Schaepman 2008; FAO 2011) are experiencing soil degradation, soil C losses, or both. In California, historical conversion from perennial to annual grasses in the 1800s has been linked to long-term soil C losses (Biswell 1956; D'Antonio et al. 2007; Ryals et al. 2014; Ryals et al. submitted). In a coastal grassland, soils under invasive annual grasses had 40 t C ha^{-1} ($147 \text{ t CO}_2\text{-eq ha}^{-1}$) less in the top 50 cm than soils under perennial grasses, suggesting that soils can lose significant amounts of C due to this plant community change over

time (Koteen, Baldocchi, and Harte 2011). Soil C losses from a valley grassland averaged 5.5 t CO₂-eq ha⁻¹ y⁻¹ over three years (Chou et al. 2008), and similar values were found in a three-year study that included a coastal and valley grassland (1.6 and 8.4 t CO₂-eq ha⁻¹ y⁻¹, respectively; Ryals et al. submitted).

Elevated atmospheric CO₂ and N fertilization can also drive changes in ecosystem C dynamics (Jones and Donnelly 2004; Sillen and Dieleman 2012). Although both elevated CO₂ and added N can increase plant productivity, these gains do not necessarily lead to increases in soil C (Sillen and Dielman 2012). In semi-arid California grasslands, three years of exposure to elevated CO₂ led to an increase in soil C but only in rapidly cycling C pools (Hungate et al. 1997). In southern California chaparral, five years of elevated CO₂ increased both soil organic and inorganic C (Treseder et al. 2003) but decreased soil organic C stability, leading to significant losses in this pool two years after treatments ceased (Trueman et al. 2009).

Anthropogenic deposition of N is a growing problem in California, particularly in chaparral and coastal scrub shrublands (Fenn et al. 2003), where deposition rates typically range from 20 to 45 kg N ha⁻¹ y⁻¹ (Meixner and Fenn 2004). In an N deposition experiment, dry-season N addition increased N in tissue and litter after one year, but did not increase soil C storage (Vourlitis, Zorba, Pasquini, and Mustard 2007). Throughout four years of N additions, C pools in the top 10 cm did not significantly change in the chaparral, but C in coastal scrub soils varied seasonally and declined slightly over time (Vourlitis and Pasquini 2009).

Some researchers have suggested that there may be an upper limit to soil C storage (e.g., Six, Conant, Paul, and Paustian 2002; Stewart et al. 2007, 2008), but this concept remains controversial, and most ecosystems are well below a hypothetical maximum storage capacity. In California, many systems are actually losing C, and field surveys indicate that most areas are well below their potential C storage capacity (Silver, Ryals, and Eviner 2010). Degraded lands may have the greatest potential for soil C sequestration (Follett and Reed 2010). However, even productive lands can continue to sequester C at high rates. For example, Gulde et al. (2008) found that agricultural soils continued to sequester over 3 t CO₂-eq ha⁻¹ y⁻¹, even after 30 years of annual manure application.

Summary of Key Findings: Soil C and CO₂ Dynamics

- Current knowledge of California rangelands indicates that these systems hold a significant amount of soil C, that soils statewide contain a wide range of soil C, and that management is a key driver of soil C content.
- California annual grasslands may be experiencing long-term C losses as a legacy of cover-type change. Many rangelands in California are experiencing widespread changes in plant cover that affect soil C pools (i.e., shifts from perennial to annual grasslands, annual grass invasion of shrublands, shrub encroachment in grasslands), and such transitions may be enhanced with climate change. Management impacts must be measured relative to baseline annual changes in soil C.
- Changes to the frequency, intensity, and timing of rainfall events are likely, and the consequences for soil C are highly uncertain. Altered rainfall patterns could potentially result in soil C losses, particularly in annual grasslands, where the majority of plants die in late spring or early summer and cannot take advantage of late or early season rain for C fixation and growth. More research is needed to understand the potentially changing baseline in soil C throughout California rangelands, which may have implications for opportunities to sequester soil C.
- Long-term research sites have been critical to identifying much of the existing knowledge on soil C pools and CO₂ fluxes under various environmental and management conditions, yet such sites are sparse and not well-replicated across California's diverse rangelands. More long-term research sites are needed to identify management strategies that enhance the C sink of rangelands under current and future climate scenarios.

Methane and Nitrous Oxide Fluxes from Rangelands

Methane (CH₄) has a global warming potential 72 times that of CO₂ over a 20-year timeframe and is 25 times more potent than CO₂ on a mass basis over 100 years (Solomon et al. 2007). Measurements of CH₄ fluxes in rangelands are relatively few, but rangelands are typically a net sink (Le Mer and Roger 2001); uptake averages -0.05 t CO₂-eq ha⁻¹ y⁻¹ but is up to -0.12 t CO₂-eq ha⁻¹ y⁻¹ in coarser-textured soils (Del Grosso et al. 2000). Overall, the sink capacity of grasslands tends to be higher than that of cultivated lands but lower than that of forests (Le Mer and Roger 2001; Aronson and Helliker 2010). When soils are saturated, rangelands can be a net source of CH₄, thus the highest uptake rates are likely at intermediate moisture levels (Dijkstra, Morgan, Follett, and LeCain 2013). Nitrogen fertilization can stimulate CH₄ uptake at low rates (<100 kg N ha⁻¹), but higher N addition can reduce the sink strength; inhibition of the CH₄ sink with N uptake appears to be greater in managed and previously fertilized soils (Aronson and Helliker 2010). In California, coastal and valley annual grasslands were net CH₄ sinks (-0.02±0.01 and -0.03±0.01 t CO₂-eq ha⁻¹ y⁻¹, respectively; Ryals and Silver 2013). Peatland soils tend to be a net source of CH₄ (Le Mer and Roger 2001), and recent measurements at the Bay Delta suggested net emissions rates of 1.1 t CO₂-eq ha⁻¹ y⁻¹ (Hatala, Detto, and Baldocchi 2012), due primarily to emissions from flooded drainage ditches and saturated regions of soils (Teh et al. 2011).

Nitrous oxide (N₂O) is approximately 289 times more powerful a greenhouse gas than CO₂ on a mass basis over 20 years and 298 times more potent over 100 years (Solomon et al. 2007). Soil N₂O emissions occur both directly (through nitrification and denitrification) and indirectly (via volatilization and redeposition or leaching and runoff and subsequent nitrification or denitrification) (De Klein et al. 2006). High temporal and spatial variability in combination with small N₂O concentrations make it difficult to quantify annual fluxes and to compare environmental and management effects (Bouwman, Boumans, and Batjes 2002; Desjardins et al. 2010). Key drivers of N₂O fluxes include substrate supply (N additions and mineralizable organic N in soils), soil moisture, and temperature (Skiba and Smith 2000). Labile C concentrations also affect N₂O fluxes as a requirement for denitrification (Rochette et al. 2010). Nitrous oxide emissions are stimulated by rainfall (Fierer and Schimel 2002; Groffman et al. 2000), but N₂O consumption by denitrifiers in mesic environments can either substantially reduce net N₂O emissions or lead to a small N₂O sink (Dijkstra, Morgan, Follett, and LeCain 2013; Chapuis-Lardy et al. 2007). A global analysis of N₂O emissions using field observations and a modeling approach found that rangelands emit N₂O at low, but non-negligible, rates (Zhuang, Lu, and Chen 2012). Savannas were the greatest emitters (0.25 t CO₂-eq ha⁻¹ y⁻¹), followed by woody savannas (0.19 t CO₂-eq ha⁻¹ y⁻¹), closed shrublands (0.12 t CO₂-eq ha⁻¹ y⁻¹), grasslands (0.10 t CO₂-eq ha⁻¹ y⁻¹), and open shrublands (0.08 t CO₂-eq ha⁻¹ y⁻¹). Valley and coastal annual grasslands from north central California emitted approximately 0.02 ± 0.02 and 0.17 ± 0.07 t CO₂-eq ha⁻¹ y⁻¹, respectively (Ryals and Silver 2013). A savanna in southern California had no significant N₂O flux during 180 days of measurements (Anderson and Poth 1989). Laboratory and short-term experiments with annual grassland soils showed strong pulses of N₂O following wet-up events (Davidson 1992; Ryals and Silver 2013). Nitrogen fertilization of annual grasslands increased N₂O emission following wet-up events, but elevated atmospheric CO₂ did not (Hungate et al. 1997). Nitrification and denitrification were both affected by several global change drivers (Barnard et al. 2006), and experimentally increased precipitation significantly increased N₂O fluxes, especially in combination with N additions and temperature increases (Brown et al. 2012).

Summary of Key Findings: Soil CH₄ and N₂O Dynamics

- Rangelands are typically a net sink for CH₄, with the exception of drained peatland pastures, which are a source of CH₄ emissions. Management practices that lead to soil compaction or increased soil moisture, or that involve N fertilization, can reduce uptake rates or increase net emissions.

- Existing datasets indicate that rangelands are typically a small source of N₂O emissions and that these emissions tend to increase with N additions, moisture pulses, and temperature increases. High-frequency measurements are particularly crucial for capturing N₂O emissions, which often occur in pulses following rainfall or management events.
- Long-term, frequent, and well-replicated measurements of N₂O and CH₄ emissions from rangeland soils are critical to quantifying the net GHG emissions or sinks from these systems under changing environmental or management conditions. Such datasets remain scarce, particularly in California.

RANGELAND MANAGEMENT: STRATEGIES AND IMPACTS

Rangeland management activities directly affect aboveground and belowground net primary productivity (NPP), litter accumulation, and nutrient cycling, all of which can affect soil GHG emission and C storage (Figure 2). Significant opportunities exist for increasing soil C storage through land management practices across a wide range of soil textures, climates, and hydrologic conditions (Smith 2008; Gilmanov et al. 2010; Ryals and Silver 2013; DeLonge, Ryals, and Silver 2013; Morgan et al. 2010; Schumen, Janzen, and Herrick 2002; Lal, Follett, Stewart, and Kimble 2007; Ogle, Conant, and Paustian 2004; Conant, Paustian, and Elliott 2001; Derner and Schuman 2007). The most effective management approaches are likely to be adaptive, changing as needed in response to changes on the landscape. This study reviews the following management approaches: grazing (physical and biological effects, grazer species), fire (suppression, planned burns), soil amendments (commercial fertilizers, manure, compost, organic matter), cultivation (mowing, irrigation, aeration and tillage), and plant community composition (species removal, species introduction).

Grazing

The effects of grazing on rangeland productivity, soil C content, and GHG emission are variable and depend on local climate, soil, topography, plant community, and grazing timing and intensity (Huntsinger, Bartolome, and D'Antonio 2007; Jackson and Bartolome 2007). In general, the animals themselves (particularly cattle) are the greatest sources of greenhouse gases from rangelands (e.g., Beauchemin et al. 2010; Eagle and Olander 2012; Stackhouse-Lawson, Rotz, Oltjen, and Mitloehner 2012). This study focuses primarily on emissions directly or indirectly from rangelands. Very little of the research on grazing management has occurred in California. Also, much of the existing research is limited by lack of replication, co-variation with other important factors (such as soil type), and short duration (Trimble and Mendel 1995). Another common limitation of published studies is the comparison of ungrazed land with overgrazed areas, ungrazed areas, or areas managed at only one grazing intensity, which fails to elucidate an optimum grazing intensity.

Grazing can compact or otherwise physically disturb soils, particularly when animal densities are high for prolonged periods and when soils are saturated (Greenwood and McKenzie 2001). Compaction generally decreases plant growth and thus C uptake from the atmosphere. It can also affect the soil hydrologic cycle by decreasing infiltration rates, lowering soil aeration, and altering the composition and diversity of soil biota (Schon, Mackay, and Minor 2012), all of which promote CH₄ and N₂O production (Reed and Petersen 1961; Asner et al. 2004). Soils vary in their degree of susceptibility to compaction and other factors, such as burrowing animals or freeze-thaw cycles, which can counteract compaction (Abdel-Magid, Trlica, and Hart 1987). Trampling by grazing animals can also destroy surface soil structure, disturb surface crusts, and create microtopography (Trimble and Mendel 1995), which can increase soil C losses through wind and water erosion. Increased runoff can enhance erosion and lead to the physical removal of C from the ecosystem, particularly in non-forested rangelands (Trimble and Mendel 1995).

Grazing directly affects the structure and function of plant communities through selective plant removal (Huntsinger, Bartolome, and D'Antonio 2007), defoliation, and changing the amount and composition of residual biomass (Bartolome, Stroud, and Heady 1980; Allred, Fuhlendorf, Smeins, and Taylor 2012; Ingram et al. 2008; Rook et al. 2004). The amount of defoliation also affects subsequent forage production by changing light competition (Collins et al. 1998) and residual biomass (e.g., Bartolome, Stroud, and Heady 1980; Phelan, Casey, and Humphreys 2013). Soil moisture decreases and temperature increases with removal of vegetation (Asner et al. 2004; Bryant et al. 1990; Bremer, Auen, Ham, and Owensby 2001), which can result in more decomposition at the soil surface and less transfer of plant litter into the soil organic matter pool. Finally, grazers redistribute nutrients through their excrement. Dung and urine patches concentrate nutrients (Hoefl, Steude, Wrage, and Veldkamp 2012) and are hotspots of organic matter deposition, with impacts on plant community composition and growth.

Several studies have found that grazing has no negative impact on soil C pools (Dahlgren, Singer, and Huang 1997; Gill 2007; Silver et al. in prep) or that it can be an integral component of healthy and productive rangelands (reviewed by Milchunas and Lauenroth 1993). Residual biomass is one of the strongest controls on rangeland productivity (Bartolome, Stroud, and Heady 1980), and thus carefully managed grazing can stimulate plant growth or reverse rangeland degradation (Dostalek and Frantik 2008). Global models of grassland soil C under different levels of grazing suggest that decreasing grazing intensity from “extreme” or “strong” to “moderate” can shift grassland soils from net sources of C to the atmosphere to net C sinks (Conant and Paustian 2002). Potential C sequestration rates from improved grazing practices are highly uncertain. A few studies have suggested that rates of up to 0.6 to 1.3 t CO₂-eq ha⁻¹ y⁻¹ may be possible (reviewed by Eagle and Olander 2012; Conant and Paustian 2002; Conant, Paustian, and Elliott 2001; Follett, Kimble and Lal 2001), while others have reported lower ranges (i.e., 0.3-1.1 t CO₂-eq ha⁻¹ y⁻¹; Derner and Schuman 2007). However, studies are scarce and some research has reported losses of up to 1 t CO₂-eq ha⁻¹ y⁻¹ due to changed grazing management (reviewed by Eagle and Olander 2012). If achievable, soil C sequestration at rates of between 0.3-1.3 t CO₂-eq ha⁻¹ y⁻¹ on 5% of California's rangelands through improved grazing could amount to a mitigation potential of 0.3 to 1.5 million t CO₂-eq y⁻¹. More research is needed to accurately predict the mitigation potential and scalability of this practice.

Livestock species also affects soil C and soil GHG emissions due to differences in animal size and behavior, manure production and composition, enteric fermentation production, and grazing preferences (Jackson and Bartolome 2007; Hoefl, Steude, Wrage, and Veldkamp 2012). Cattle exert enormous pressure on the soil surface and prefer riparian areas to uplands, which can erode stream banks and lead to local soil C losses (Trimble and Mendel 1995). Goats are better browsers than sheep and cattle and can consume relatively more shrubs (Rogosec, Pfister, Provenza, and Grbesa 2006; Salem, Salem, El-Adawy, and Robinson 2006; Alonso-Diaz, Torres-Acosta, Sandoval-Castro, and Hoste 2010), affecting plant communities and related C stocks (discussed below). Horses also increase soil compaction and alter plant community composition (Beever, Tausch, and Brussard 2003). Breeds within all species can exhibit different behaviors and impacts, thus animals may be selected or bred for traits that minimize GHG emissions (Estell et al. 2012).

Most rangelands in California are grazed, thus grazing management has a large potential to sequester C and minimize GHG emissions. However, existing research is limited, particularly in California, and results are highly uncertain. Long-term, well-replicated research studies are needed to identify best grazing management practices.

Fire

California rangelands, like most rangelands, have a long history of natural (lightning-induced) and anthropogenic fire (Pyne, Andrews, and Laven 1996; Fuhlendorf et al. 2012). Native Americans used fire

to manage rangelands beginning about 10,000 years ago and continuing until the fire suppression efforts of the late 19th century (Reiner 2007). California chaparral and coastal scrub are prone to fire and typically burn every few decades (Keeley 2002; Minnich and Bahre 1995). Prescribed fires have been used as a management tool to clear aboveground biomass, control woody biomass (Bond and Keeley 2005), maintain native species and biodiversity (Anderson 2006; Fuhlendorf, Engle, Kerby, and Hamilton 2009; Reinkensmeyer, Miller, Anthony, and Marr 2007), and add C and N to the soil as charcoal (Collins et al. 1998; Knicker 2007; Reiner 2007 and references therein; Bremer and Ham 2010). For a variety of reasons, approximately 60,000 ha of wildlands are annually treated with prescribed burning; increasing risks of wildfires means that this treated land area is projected to grow (<http://www.arb.ca.gov/smp/progdev/pubeduc/pbfs.pdf>).

Greenhouse gas emissions from the combustion of vegetation are significant (Simpson et al. 2006), but measurements from temperate rangelands are scarce (Urbanski 2013; Yokelson et al. 2013). Although the C released as CO₂ is of recent origin and likely to be reabsorbed during regrowth, emissions of numerous other aerosols and gases contribute to global warming (Smith et al. 2008; Simpson, Rowland, Meinardi, and Blake 2006). Recent work suggests emissions factors for rangelands of approximately 0.06 t CO₂-eq t⁻¹ dry biomass for CH₄ and 0.09 t CO₂-eq t⁻¹ dry biomass for N₂O (Urbanski, Salmon, Nordgren, and Hao 2009). However, these values are highly uncertain and vary substantially depending on fire type. Using estimates of dry biomass of some rangeland types in California and including CH₄ and N₂O only, these emissions factors can be scaled to an area basis. Annual grasslands with 0.5–2.5 t standing biomass ha⁻¹ (Jackson and Bartolome 2007) could produce up to 0.1–0.4 t CO₂-eq ha⁻¹ from a burning event. Southern California chaparral, with 16–52 t biomass ha⁻¹ (Hardy, Conard, Regelbrugge, and Teesdale 1996), could produce up to 2.5–7.8 t CO₂-eq ha⁻¹ from a burn event. Although CO₂ emissions from fires are generally not considered to contribute to global warming (Smith et al. 2008), fires can affect ecosystem C stocks in the short-term during the recovery period or in the long-term if the plant community changes. In a tallgrass prairie system, CO₂ emissions from fire led to a net loss of C over 2 years of measurements despite increases in aboveground NPP (Bremer and Ham 2010). Fire combusts near-surface soil organic matter through oxidation (Knicker 2007). Biochar, the residue from incomplete combustion can be resistant to decomposition in some soils, increasing C in stable C pools in some (Rice and Owensby 2001) but not all cases (Knicker 2007; Suyker and Verma 2001, but see Schmidt et al. 2011).

Indirect effects of fire on soil organic matter and GHG emission are related to changes in nutrient cycling and plant composition. Fire may favor perennials in systems where annual grass seeds are fire intolerant (Reiner 2007). Fire that is too frequent can limit NPP due to incomplete recovery between disturbances and the volatile loss of C and N that would otherwise be incorporated into the soil (Reiner 2007; Cook et al. 2010). The volatile loss of N may promote native species in grasslands affected by high N deposition (Reiner 2007). However, controlled burns were ineffective in promoting natives and caused numerous detrimental effects in a California coastal scrub ecosystem (Meixner et al. 2006). One of the challenges in managing rangeland fire is the need to address multiple objectives, including minimizing GHG emissions, promoting biodiversity, controlling invasive plants, protecting human infrastructure and life, abating woodland encroachment, and increasing ecosystem productivity.

Fire is often used in rangelands to control aboveground biomass and plant community composition, which can indirectly affect soil C stocks. Fires also emit CH₄, N₂O, and other greenhouse gases in amounts that are poorly characterized due to lack of research. Management approaches using fire should consider both the long-term impacts on soil C as well as GHG emissions during burning.

Soil Amendments

Soil amendments can be added to rangelands to increase NPP or to alter soil nutrient stocks and dynamics. In 2007, 261,145 ha of California's total land area received manure; an additional 59,000 ha of pastureland and rangeland received commercial fertilizers, lime, and other soil conditioners (representing only 2.2% of the total agricultural acreage treated; USDA 2007). Nitrogen limitation is common in many rangelands and is likely to be important in California (Haubensak and D'Antonio 2011). Thus, amendments rich in N often stimulate NPP (Harpole, Goldstein, and Aicher 2007; Harpole, Potts, and Suding 2007; Derner and Schuman 2007). Fertilization can also increase soil C sequestration, with reported rates ranging from 0.4 to 5.9 t CO₂-eq ha⁻¹ y⁻¹ (Eagle and Olander 2012). Improved efficiency in the use of nutrients (mainly N, but also P) on grazing lands can minimize direct and indirect GHG emissions from soils and animal manure, fertilizer production and application, and feed cultivation (Bolan et al. 2004). Organic amendments need to be tested and monitored, because they can contain metals, metalloids, and other toxins that could contaminate soils if added in large quantities or in frequent applications (Park et al. 2011). However, improvements in wastewater treatment technologies and in feed utilization in animal industries are decreasing contamination in organic waste materials to the point that these materials can often even be used to remediate contaminated soils (Park et al. 2011).

Organic matter additions can increase both NPP and soil C storage in grasslands (Paustian, Collins, and Paul 1997; Conant, Paustian, and Elliott 2011; Lal 2004a,b; Smith et al. 2008; Cabrera et al. 2009; Ryals and Silver 2013; Albaladejo et al. 2008; Eagle and Olander 2012). Organic amendments also increase soil fertility, soil water-holding capacity, and drought resistance (Hudson 1994; Diacono and Montemurro 2011). The trend in animal agriculture toward production in confinement facilities is resulting in the localized concentration of manure in excess of agricultural soil nutrient requirements (Kellogg, Lander, Moffitt, and Gollehon 2000). Application of this manure to rangelands offers a means of redistributing the nutrients, decreasing nutrient pollution, and increasing forage production (Kellogg, Lander, Moffitt, and Gollehon 2000). Composted or anaerobic digestate of organic wastes, such as dairy slurry, urban wastewater, and food processing waste, can be used as an organic matter addition to soils without loss in productivity (Walsh et al. 2012) and is likely to reduce net GHG emissions (DeLonge, Ryals, and Silver 2013).

Compost may be particularly effective at sequestering C (Ryals and Silver 2013; Fronning, Thelen, and Min 2008). In California coastal and valley grasslands, compost C directly increased ecosystem C by 52 t CO₂-eq ha⁻¹ and also enhanced NPP by 2.6 and 5.7 t CO₂-eq ha⁻¹ y⁻¹, respectively, during three years following application; the consequential C sequestration rate was estimated to be 0.6 to 4.1 t CO₂-eq ha⁻¹ y⁻¹ (Ryals and Silver 2013). Modeling using DayCent indicated that these effects could persist for decades (Ryals et al. submitted). Scaled up to 5% of California's rangeland, such sequestration rates could potentially mitigate 0.7 to 4.7 million t CO₂-eq y⁻¹. Diversion of waste stream materials for compost production can add climate change mitigation value to this management approach. A life-cycle assessment (LCA) case study for this practice based on compost produced from materials diverted from high-emission waste systems (liquid manure storage and landfills) indicated that 23 t CO₂-eq ha⁻¹ could be saved within three years following a one-time application of compost at 250 kg N ha⁻¹ (DeLonge, Ryals, and Silver 2013). The LCA assumed that increased forage production led to reduced demand for commercial feed as well as a modest annual increase in soil C storage, but most mitigation potential in this case was from improvements in waste management; emissions from all phases of the life cycle were considered, including transportation and application. An uncertainty analysis using a broader range of variables suggested a net mitigation potential of 4.3 ± 0.8 t CO₂-eq ha⁻¹ would be likely (DeLonge, Ryals, and Silver 2013). Using the more conservative estimates from the uncertainty analysis, 4.9 million t CO₂-eq could be mitigated annually if this practice was applied to 1.1 million ha (5%) of California's rangeland each year. Research identifying the locations and availability of organic waste and improving emissions estimates from the composting process is needed to improve estimates of the mitigation

potential of this management approach. Field research on diverse rangelands, particularly more arid systems, is required to evaluate the impacts of scaling up this approach.

The impacts of soil amendments on soil C and soil GHG emissions depend on the physical and chemical properties of the amendments. Mineral fertilizers can enhance NPP but may also inhibit CH₄ oxidation and thus reduce CH₄ uptake by up to 75% (Del Grosso et al. 2000), although not in all cases (Bodelier and Laanbroek 2004). Mineral N fertilizers often increase N₂O emissions (Mosier et al. 1991). Additionally, greenhouse gases are emitted during the production of inorganic N fertilizers, at rates dependent on the N form and the production process. Typical emissions rates for ammonium nitrate production are 3.0–7.1 t CO₂-eq t⁻¹ N and for urea are 0.9–4.0 t CO₂-eq t⁻¹ N (Wood and Cowie 2004). Organic mulches (straw, bark) have been used to reduce N availability in soils with high rates of N deposition in southern California; this management practice has been applied in disturbed coastal scrub with the goal of preventing further annual grass invasion (Allen and Zinc 1998). The mulch increased microbial activity, immobilized N, improved the survival rate and productivity of planted native perennials, and led to increased soil organic matter in the amended plots after one year (Allen and Zinc 1998).

Nitrous oxide emissions tend to be elevated in grasslands that receive large amounts of manure or N fertilizer additions; estimates suggest that 2.5% of synthetic-N fertilizer and 2.0% of manure-N is ultimately converted to N₂O globally (Davidson 2009; Mosier et al. 1996; Bolan et al. 2004). Depending on the form added, organic matter applications may increase GHG emissions from rangeland, particularly of N₂O (DeLonge and Silver in prep; Owen and Silver in prep). However, research suggests that emissions may be minimized by using composted materials, which act as slower-release fertilizers (Ryals and Silver 2013; Dalal, Gibson, and Menzies 2009; Dalal, Gibson, Allen, and Menzies 2010; Paul, Beauchamp, Zhang, and Zhang 1993; Eghball 2000; Sikora and Szmids 2001). Compost complexes N; thus, it is an organically-bound, slow-release form of N as well as a source of recalcitrant C potentially unavailable to soil denitrifiers. This property of compost may increase N immobilization rates, slow N mineralization rates, and decrease N₂O fluxes (Huang et al. 2004), while simultaneously improving long-term soil fertility by building soil organic matter. Enhanced-efficiency N fertilizers (EENFs) also have the potential to decrease GHG emissions relative to conventional fertilizers (Halvorson et al. 2013; Akiyama et al. 2010). These EENFs include stabilized EENFs (containing nitrification, urease inhibitors, or both), slow-release EENFs (which contain slowly released N components with variable release rates), and controlled-release EENFs (which have more predictable release rates) (Halvorson et al. 2013). EENFs have been found to reduce N₂O in cropping systems by 14% to 61% relative to commonly used synthetic N fertilizers (Halvorson et al. 2013). Efficient soil amendments that minimize GHG emissions while enhancing NPP can help to mitigate climate change.

Soil amendments can improve rangeland productivity and sequester soil C, while providing numerous co-benefits. Thus this approach is promising and a high research priority. Slow-release and high-efficiency fertilizers can maximize benefits while minimizing soil N₂O losses and transportation costs. When composts are produced from materials diverted from high-emission waste streams, an additional benefit is reductions in emissions from the waste sector. This approach must be considered in a full life-cycle assessment context.

Cultivation

Mowing

Mechanical mowing can have similar impacts as grazing events on soil C and GHG emissions, because both reduce biomass. Like overgrazing, equipment used for mowing can compact soils. Energy used for mowing leads to GHG emissions. Mowing events can reduce C uptake (Barcza et al. 2003), but they did

not explain differences in annual NEE over six years in a montane grassland (Wohlfahrt et al. 2008). At subalpine grassland sites, grass cutting reduced C uptake by approximately 50%, and the cut meadow was a net source of C during the growing season ($2.9 \text{ t CO}_2\text{-eq ha}^{-1}$), whereas a neighboring undisturbed meadow was a C sink ($-4.8 \text{ t CO}_2\text{-eq ha}^{-1}$) (Rogiers, Eugster, Furger, and Siegwolf 2005). In mowing studies, biomass has often been removed from the site and decomposed or consumed elsewhere; emissions from the decomposition of this biomass have not usually been quantified (Barcza et al. 2003). In some cases, mowing alters the plant community composition; this change may have a greater impact on net C exchange than the loss of biomass. For example, when pepperweed was mowed from a drained peatland pasture in California, reduced energy reflectance and increased photosynthetic capacity led to a net C sink relative to other periods (-6.2 vs. -1.1 to $4.8 \text{ t CO}_2\text{-eq ha}^{-1} \text{ period}^{-1}$; Sonnentag et al. 2011).

Aeration and Tillage

Aeration and tillage are management strategies that mechanically agitate soils and can counteract soil compaction. Aeration can reduce woody plant cover while promoting grasses (Fulbright and Ortega-Santos 2013). Disturbances associated with aeration and tillage can sometimes enable exotic grass invasion (Ayala-A et al. 2012). Aeration may promote NPP in some cases, but it also disrupts surface soils and breaks up soil aggregates that protect soil C, promoting soil C losses as well as erosion (Gebhardt, Daniel, Schweizer, and Allmaras 1985).

Tillage is applied primarily to croplands, where it has been well studied (Eagle et al. 2010), but it is also occasionally used on rangeland systems, where it has been less well studied. Conventional plowing results in significant losses of soil organic C across agricultural lands (Reicosky 2003; Lal 1993). Plowing also requires fossil fuel consumption (Philips et al. 1980). Research has suggested that less intensive methods, such as conservation tillage, can minimize C losses relative to conventional plowing (West and Post 2002), although gains in surface soils may be offset by losses deeper in the soil profile (Baker, Ochsner, Venterea, and Griffis 2007). In California, mechanical disturbances may have led to permanent conversion of some coastal scrub to annual grasslands; this conversion is possibly related to the disruption of arbuscular mycorrhizal fungi (Engelberg et al. 2013).

Irrigation

Rangeland in California is largely not irrigated. In 2008, irrigation occurred on 119,655 ha (6%) of cropland used for pasture or grazing and 29,028 ha (3%) of pastureland and rangeland (USDA 2008). These values are somewhat different than those for the entire United States, where less than 2% of pastureland and rangeland were irrigated, but more than 15% of cropland used only for pasture or grazing were irrigated (USDA 2008). Irrigation has been used to increase NPP in drylands, but few studies have investigated the net ecosystem C impacts (Eagle and Olander 2012). Irrigation led to an increase in soil C pools of $0.7 \text{ t CO}_2\text{-eq ha}^{-1} \text{ y}^{-1}$ in Australian rangelands (Rixon 1966) and an increase of approximately $1.8 \text{ t CO}_2\text{-eq ha}^{-1} \text{ y}^{-1}$ in Idaho, but not in a New Zealand study (Houlbrooke, Littlejohn, Morton, and Paton 2008). Although irrigation is desirable for forage production, it can decrease soil C storage by increasing decomposition rates relative to C inputs (Kelliher, Condon, Cook, and Black 2012) and increase N_2O emissions (Chou et al. 2008). The timing of irrigation is particularly important to GHG production. Irrigation can lead to nutrient losses through runoff and leaching when poorly timed (Bush and Austin 2001). With climate change, more irrigation may be needed to offset soil moisture deficits associated with increasing frequency and severity of drought (Mote, Hamlet, Clark, and Lettenmaier 2005) and higher temperatures (Cayan et al. 2008).

Mowing, aeration, and tillage disturb rangeland soils, typically leading to soil C losses. Irrigation can increase productivity and therefore soil C, but water shortages limit the practical application of this

approach. All of these approaches also require significant energy to implement, minimizing the net mitigation potential.

Plant Community Composition

Management of plant community composition includes seeding, invasive plant control, woody plant management, and plant breeding. On pasturelands, ranchers or land managers may spread seed to increase NPP and the nutritional value of forage (Abberton et al. 2008). Increases in productivity generally result in increased soil C content (Conant, Paustian, and Elliott 2001). If legume seeds are added, the increase in N fixation can stimulate plant productivity and soil C sequestration (Watson 1963). For example, interseeding mixed-grass rangelands in the Great Plains with a legume led to an increase in NPP and a corresponding increase in soil organic C by 4% just three years after interseeding and by 17% after 36 years (Mortenson et al. 2004, 2005). Legumes can also decrease the need for N fertilizers, reducing the loss of excess N to the atmosphere as N₂O or to groundwater (Abberton et al. 2008).

In California, the replacement of native perennial with exotic annual grasslands may decrease soil C storage (Koteen, Baldocchi, and Harte 2011) and increase rates of N cycling and associated losses (Parker and Schimel 2010). Grassland soil aggregates were less stable under non-native plants than native ones, possibly due to differences in roots or microbial community (Duchicela et al. 2012), which could decrease the mean residence time of C and N in soils. Plant composition management may be challenging in some areas of California where high N deposition is driving invasive plant expansion, in addition to increasing soil N₂O emissions (Fenn et al. 2003). The critical deposition load at which invasive species are favored in these environments is between 3–8 kg N ha⁻¹ y⁻¹, and 10% of California's land area already receives >10 kg N ha⁻¹ y⁻¹ (Fenn et al. 2011, 2010). Plant management through burning emits greenhouse gases, as described above (Urbanski 2013). As an alternative to burning, weeding has been proposed as a means of controlling invasive plants (MacDougall and Turkington 2007), but it is cost-prohibitive for large areas, and the effects on ecosystem C remain uncertain for most rangelands.

Woody plant encroachment, the replacement of rangeland grasses with shrubs or trees, is particularly common in arid and semi-arid ecosystems (Van Auken 2009; Asner et al. 2004) due to climatic variations, overgrazing, changing fire regimes, and increased atmospheric CO₂ (Archer, Schimel, and Holland 1995; Barger et al. 2011; Roques, O'Connor, and Watkinson 2001; Gao and Reynolds 2003; Grice 2006; Morgan et al. 2007; Van Auken 2009). The growth of woody plants may result in a net ecosystem gain or loss of C and N, depending on temperature and precipitation (Jobbágy and Jackson 2000; Jackson et al. 2002; Barger et al. 2011). With woody encroachment, soil organic C may decrease (Jobbágy and Jackson 2000), while aboveground C stocks increase (Asner et al. 2004). As a result of woody encroachment, rangeland in the San Joaquin River Valley and Mojave Desert (where mean annual precipitation is less than 336 mm) would likely experience reduced aboveground NPP, whereas other rangeland in California would likely have increased aboveground NPP (Barger et al. 2011).

California oak woodlands represent a unique case of woody plant growth in rangelands. Oak woodlands, with an annual grass understory, are the natural cover for the hills surrounding the Central Valley but are under threat from development and unknown factors limiting oak seedling success (Reiner and Craig 2011). Oak removal was once recommended as a means to improve rangelands by reducing competition for moisture (Lewis 1968), but this potential benefit was short-lived (Kay 1987). Research has shown that soils under oak understories store more C and N and exhibit more favorable soil conditions than deforested areas (Dahlgren, Singer, and Huang 1997). Although forage production is similar in oak savannas and open grasslands, soil organic N and C could be lost following oak removal (Jackson, Strauss, Firestone, and Bartolome 1990). In areas still populated by oaks, grazing may decrease oak seedling recruitment; however, that effect is difficult to separate from the effects of interannual precipitation variation and competition from annual grasses (Reiner and Craig 2011).

Targeted plant breeding or species selection has been proposed to increase the sustainability of rangeland-based industries. Breeding can be used to improve the sustainability of agriculture (as reviewed by Humphreys et al. 2006). This practice could decrease the need for N fertilizers by increasing N-use efficiency or soil C sequestration, or by improving digestibility and decreasing enteric emissions (Abberton et al. 2008, and references therein). Forage composition is well known to affect enteric emission; a variety of feeding strategies are being investigated to reduce CH₄ emissions (Beauchemin et al. 2010). Changing protein and energy content of forage may increase N-use efficiency in the rumen and decrease N excretion and subsequent conversion into N₂O (Humphreys et al. 2006). Shrubs might also be bred to be more palatable to grazers by reducing secondary metabolites, enabling increased grazing in areas experiencing woody plant encroachment (Estell et al. 2012). Secondary or unexpected consequences of plant introductions can have deleterious effects on ecosystem C storage, hydrology, and biodiversity; thus, research is needed before such activities could be recommended in practice.

Plant communities affect soil C and GHG emission on rangelands; thus, plant community management may contribute to climate change mitigation. Opportunities for such management are closely related to fire and grazing and are likely to be site-specific. Interseeding of legumes in rangelands and plant breeding are promising approaches that require additional research for California.

Summary of Key Findings: Rangeland Management

- Grazing directly affects rangeland soils and plants, but the effects of grazing on soil C pools and GHG emissions are poorly understood. Improved grazing practices (i.e., lowered or managed grazing intensity) may increase C storage in soils.
- Research on greenhouse gases and C exchange from grazing systems has been conducted primarily on annual grasslands and is sparse on other types of rangeland systems, particularly in more arid regions of California.
- Soil amendments can increase nutrient availability and NPP, sequestering atmospheric CO₂. Organic matter amendments provide numerous co-benefits. Slow-release fertilizers, such as compost, may facilitate C uptake for decades without additional application or maintenance. Amendments can have different impacts, depending on their chemical and physical properties. For example, increasing soil N availability can enhance the productivity and soil C stocks of annual grasslands, whereas immobilizing soil N may improve productivity and soil C in some coastal scrub ecosystems.
- In some cases, soil amendments can promote N₂O emission and reduce the CH₄ uptake of rangelands, particularly when soils are saturated. Timing, quantity, and quality of amendments can minimize or eliminate these changes to GHG fluxes.
- Compost amendments may be particularly effective at sequestering soil C while minimizing increased N₂O emissions or decreased CH₄ uptake in rangeland soils. An added benefit of compost is that it can be produced from manure and plant materials that typically emit large amounts of CH₄ when stored in common waste management systems such as anaerobic lagoons and landfills. Thus, compost applications can reduce emissions from the waste management sector.
- Mowing or mechanically harvesting plant material from pastures leads to immediate reductions in C uptake that do not appear to be compensated for during re-growth, leading to net reductions in C storage. These practices also require energy consumption for equipment operation.
- Plowing increases soil C losses and can disrupt microbial and fungal communities, leading to long-term or permanent changes in the ecosystem.

- Irrigation can increase soil C storage and minimize losses during droughts, but increased soil moisture after dry periods and during the summer can enhance soil GHG emissions.
- Plant communities play a large role in soil C and GHG emissions and are frequently in transition in California's rangelands due to climate variability, fire, invasive species, and land use change. Species removal and introduction have a history of decreasing soil C storage in California rangelands.
- Rangeland systems store more C in soils and produce fewer GHG emissions than croplands or urban land uses. Preventing conversion of rangelands to alternate systems can help to mitigate climate change.

TOOLS FOR MANAGEMENT DECISIONS: MODELING AND REMOTE SENSING

Ecosystem and biogeochemical models are valuable tools to be used in conjunction with field research to predict changes in soil C pools and GHG emissions over broad spatial and temporal scales and at resolutions not feasible in the field. Life-cycle assessments can be used to quantify and compare the overall impacts of available land management options, to identify the most critical gaps in knowledge, and to evaluate the impacts of uncertainties. Remote sensing can be used to assess rangeland conditions over large spatial and temporal scales and may be useful in identifying optimal project areas and resource allocations. Several tools will be helpful in assessing baseline conditions and identifying and implementing mitigation opportunities on California's rangelands.

Ecosystem and Biogeochemical Models

Numerous mechanistic and empirical models have been developed to investigate rangeland NPP and management, but relatively few of these models include soil C or GHG emissions (see Deneff et al. 2012; Derner, Augustine, Ascoug, and Ahuja 2012). Biogeochemical models explicitly include soil C pools and soil trace gas emissions, and some can model their dynamics in response to different management conditions. The CENTURY model was first developed for grasslands and has been used to model a variety of ecosystems, climates, and management practices (Parton, Ojima, and Schimel 1994; Kelly et al. 2000; Del Grosso et al. 2002; Parton, Morgan, Wang, and Del Grosso 2007). The DayCent model, a daily timestep version of CENTURY, accurately calculated soil C pools and GHG emissions from California chaparral, though improvements were needed for the effects of fire (Li et al. 2006). In this study, NPP and soil N fluxes exhibited high inter- and intraannual variability with large decreases in NPP and increases in soil gaseous N emissions for three years post-fire; soil C and N losses were smaller but lasted for decades. Furthermore, the modeling results suggested that increases in N deposition would lead to increased N leaching and gaseous emissions (Li et al. 2006). The DayCent model was also evaluated for two grazed annual grasslands in California: one that received compost additions and one left untreated; soil C and GHG fluxes were all within the range of observed values (Ryals et al. submitted). This study indicated that compost additions to California rangelands led to rapid increases in soil C pools and a net sink in greenhouse gases that lasted for decades (1.3–1.6 t CO₂-eq ha⁻¹ y⁻¹ over the initial 10 years; Ryals et al. submitted). Another version of the CENTURY model, MC1, was parameterized for California; simulations of large-scale climate change effects on vegetation, soil C, and fire regimes in historical time periods compared favorably with observations (Lenihan, Bachelet, Neilson, and Drapek 2008). Research using this model predicted that California grasslands will expand and that fire burn area will increase, but the net C flux from California's land mass was uncertain. A cooler, moister climate change scenario led to an annual statewide sink of 1,177 million t CO₂-eq; warmer and drier scenarios led to annual emissions of 279–473 million t CO₂-eq (Lenihan, Bachelet, Neilson, and Drapek 2008).

The Denitrification-Decomposition (DNDC) model is similar to the DayCent model but was designed specifically for agricultural ecosystems (Li 2000). Despite DNDC's similarities to the DayCent model,

fewer studies have used DNDC in rangeland systems globally or in California. DayCent was found to reproduce N₂O fluxes better than DNDC on a humid pasture (Abdalla et al. 2010). Five process-based ecosystem models (i.e., BEPS, Biome-BGC, CABLE, ORCHIDEE, JULES) did a poor job of predicting CO₂ exchange over seven Mediterranean oak woodlands during droughts; they underestimated variability over weeks to months, overestimated variability on a yearly scale, and consistently underestimated ecosystem respiration during drought (Vargas et al. 2013). Drought conditions are likely to become more common due to climate change (Kharin, Zwiers, Zhang, and Hegerl 2007).

Numerous models that have not included soil C or GHG dynamics could be leveraged to meet additional needs. Bioclimatic envelope models have historically been used to predict changes to plant communities under changing climates (Abatzoglou and Kolden 2011). These models need to be coupled with mechanistic models to link changes in plant communities with soil C pools and GHG emissions under diverse climate and disturbance regimes. Several models have been developed specifically to evaluate grazing (Tietjen and Jeltsch 2007; Wiegand, Wiegand, and Putz 2008; McKeon et al. 2009), though these models have focused primarily on forage production impacts, rather than soil C or GHG fluxes.

Some needed improvements in models include designs that facilitate decision-making (considering tradeoffs and available resources) and explicit consideration of interactions among weather, herbivory, and fire (Derner et al. 2012). Additionally, models need to be able to better predict the implications of short- and long-term climate variability for rangeland management (Teague et al. 2009; Ebrahimi, Milotic, and Hoffmann 2010).

Life-Cycle Assessments

Rangeland management decisions require consideration of the net impacts of changes in ecosystem C pools, direct rangeland emissions, and indirect emissions from management activities (Schlesinger 1999). Life-cycle assessments are a valuable tool to compare the relative importance of energy and material inputs and to assess the net effects of management on C and GHG dynamics. For example, organic matter amendments can sequester C in soils, increase forage production, reduce demand for energy-intensive imported animal feed and synthetic fertilizers, and divert materials from high-emission waste management facilities (e.g., landfills, anaerobic lagoons). However, producing these amendments, transporting them, and making subsequent dietary changes for livestock all produce GHG emissions. In one LCA, producing and adding organic amendments to soils led to a net GHG savings averaging 4 t CO₂-eq ha⁻¹, and up to 23 t CO₂-eq ha⁻¹ in one case study, due primarily to diversion of waste from high-emission manure storage systems and landfills (DeLonge, Ryals, and Silver 2013). In comparison, applications of manure from liquid storage systems and synthetic N fertilizers yielded net GHG sources (DeLonge, Ryals, and Silver 2013).

Life-cycle assessments can address the GHG impacts of a broad range of practices, such as irrigation, plant community management (through weed control, chemical additives, or fire), livestock and dairy production, manure management, and wool production. For example, energy required to pump water for irrigation emits approximately 0.7–2.9 t CO₂-eq ha⁻¹, offsetting potential gains in soil C from increased NPP, particularly in arid regions (Schlesinger 1999). Beef production LCAs have shown that large portions of total GHG emissions are from enteric emissions (> 50%) and manure (25%), but these studies have typically excluded impacts on soil C (Beauchemin et al. 2010; Stackhouse-Lawson, Rotz, Oltjen, and Mitloehner 2012). An LCA of milk production in California indicated that approximately 0.5–0.6 t CO₂-eq are produced per ton milk, but this assessment did not allow for depletion or sequestration of soil C stocks that likely occur with different land management systems (Rotz, Montes, and Chianese 2010). Research on the impacts of sheep systems on greenhouse gases remains limited (Zervas and Tsiplakou 2012). One LCA for sheep found that one ton of their meat has a GHG cost of 5 to 6 t CO₂-eq (Biswas 2010). This cost is lower than the GHG cost of beef (11 to 23 t CO₂-eq t⁻¹; Stackhouse-Lawson, Rotz,

Oltjen, and Mitloehner 2012), because sheep are smaller than beef cows and have lower rates of enteric fermentation. Wool production has been estimated at 15–17 t CO₂-eq t⁻¹ (Biswas 2010); this cost has not been compared to that for other fibers.

One of the most complex aspects of LCAs is the treatment of land use and land cover change. In many cases, C storage and GHG fluxes on rangelands are excluded from analyses (Rotz, Montes, and Chianese 2010). When included, different assumptions and treatments of land use can lead to conflicting conclusions about the GHG ramifications of products or production systems. For example, a study of milk production showed that compared with conventional systems, organic milk systems might reduce emissions by 40% or increase emissions by up to 50%, depending on the representation of land use change (Flysjo, Cederberg, Henriksson, and Ledgard 2012). In theory, a full LCA would consider all indirect land use changes as well, though these changes can be difficult to quantify.

Remote Sensing

Remote sensing can be used to assess current and past rangeland productivity, vegetation characteristics, and land management. Such information can be useful for estimating baseline rangeland productivity, evaluating the impacts of management changes, or identifying lands that might be most suitable for specific management practices. Remote sensing tools for assessing rangeland soil C and GHG dynamics are currently limited, but recent improvements are promising. In one study, available eddy covariance data from AmeriFlux was scaled up to the conterminous United States using MODIS data; the model did a good job of predicting NEE, including NEE of rangelands in California's Mediterranean climate (Xiao et al. 2008). Another study using remote sensing showed that one index explained 49% of variance in the water vapor flux in California chaparral and could be useful in detecting drought conditions, when these rangelands are particularly prone to C loss (Claudio et al. 2006). Remote sensing and modeling were combined to calculate Estimated Ecosystem Performance (EEP), which can be used to detangle influences of weather and site potential from management-induced changes (Rigge et al. 2013). Although these remote sensing methods do not quantify soil C, they have been used to effectively identify overgrazed or degrading areas in the western United States (Rigge et al. 2013). Unmanned aerial vehicles (UAVs) also have a great deal of promise for quantifying vegetation cover and rangeland conditions using remote sensing at a variety of spatial scales, particularly over rugged or remote terrain (Rango et al. 2009). Applying such techniques to California may be useful in identifying rangelands where changes to management practices are most needed or would be most fruitful.

KEY FINDINGS AND RESEARCH PRIORITIES

California's rangelands are extensive; thus, even small rates of C sequestration and emissions reduction across these landscapes have the potential to make significant contributions to the state's climate change mitigation goals. The material reviewed above highlights several avenues with considerable potential for mitigating GHG emissions on California rangelands.

Key Findings

- A large proportion of California's rangelands are likely to be degraded with regard to soil C pools and have significant potential for increased C sequestration in soils through management. Some California rangelands are experiencing C loss due to environmental or management factors that have led to shifts in vegetation; research suggests that C losses in these rangelands is likely to be at least partially reversible.

- Research suggests that organic matter additions to rangelands, particularly with composted organic wastes, may be a viable strategy for C sequestration in California's Mediterranean climate. This management approach has the additional potential benefit of GHG mitigation in several industries, including dairy, livestock, crop agriculture, and waste management.
- Well-managed grazing is not likely to decrease soil C pools on rangelands and could increase C storage. Sustainable grazing practices can increase forage per land area in the short- and long-term, reducing reliance on other animal feed products.
- Carbon and GHG emissions in California's rangelands are sensitive to changes in temperature, precipitation, management, atmospheric CO₂ concentrations, and N deposition, all of which are occurring throughout California and are likely to persist or even intensify in the near future. These environmental drivers are altering baseline soil C and soil GHG dynamics, likely with implications for management. Monitoring shifting baselines in rangelands will be critical to identifying and selecting the best management opportunities.

Research Priorities

- More research is needed to ensure that soil C stocks, net ecosystem exchange, and GHG fluxes (CH₄ and N₂O) are well understood across the full range of California rangeland cover types, including grasslands and shrublands along elevation and precipitation gradients. Carbon sequestration, avoiding soil C losses, and minimizing GHG emissions can all be part of a climate change mitigation strategy. A more complete understanding of current C pools and GHG fluxes will facilitate state-level emissions inventories and emissions reduction planning.
- Investment in long-term, well-replicated research sites is needed to accurately quantify baseline conditions on California's rangelands and to identify opportunities for management. Long-term studies are of particular importance in rangeland systems. These systems experience large interannual rainfall variability and are highly susceptible to disturbances, particularly droughts and fires, which can have long-lasting effects.
- Continuous measurements of GHG fluxes are required to accurately capture net annual emissions or sequestration. Such measurements are particularly needed for N₂O, which is often emitted in short pulses following rain events. Eddy-covariance systems with high-frequency gas analyzers will be instrumental in measuring semi-continuous GHG fluxes from whole ecosystems over spatial scales of hundreds of meters to several kilometers. Plot-scale field experiments will be needed to quantify spatial heterogeneity, identify species-specific effects, and test the effects of different management strategies.
- More research is needed on grazing management to reduce GHG emissions and increase C storage. The majority of California's rangelands are grazed. Grazing practices that rehabilitate degraded soils and increase soil C storage are needed to help the state meet its GHG emissions reduction goals, but these practices must first be identified and optimized for local conditions and needs. This research should specifically consider soil, plant, and animal C and GHG fluxes, plant community and water resources, and social issues and economic analyses associated with changing grazing regimes. Research should be conducted at a field scale with full replication and carefully selected controls. Ideally, it would be conducted in Mediterranean, arid, and semi-arid bioclimatic zones to account for some of the important diversity of rangeland cover types in the state. Empirical research should be coupled with modeling studies to facilitate scenario planning and broad-scale quantification of outcomes.

- Research on the use of organic matter amendments for climate change mitigation should be expanded to include a broader set of rangeland types, particularly rangelands in arid and semi-arid climates, and a wider range of materials. Research should include well-integrated empirical and modeling studies, and, ideally, social science and economic analyses to ensure feasibility of management alternatives.
- The interactions of grazing and fire management should be explored to determine the best management practices to meet the needs of livestock industries and fire control while reducing GHG emissions.
- Opportunities for managing rangeland plant communities to sequester soil C require greater exploration. For example, interseeding legumes has shown strong potential for building soil C in the Great Plains.
- Modeling studies should explicitly address the effects of management alternatives (including those outlined above) under changing climate scenarios. Specifically, models should explore the effects of drought, changes in the timing of rainfall, reduced snowpack, increased frequency or intensity of fire, and increased population and food demands.
- Life cycle assessments (LCAs) should be conducted for all management strategies to ensure that only strategies with a net mitigation potential in the long term are pursued. LCAs should be parameterized for local conditions whenever possible and should include statistically robust uncertainty analyses.
- Remote sensing, in combination with unmanned aerial vehicles, show tremendous promise as relatively low-cost ways to monitor rangeland conditions and plant communities, but these technologies require greater investment. Research is needed to develop algorithms linking remotely sensed data to rangeland soil C and GHG dynamics. Remote sensing should also be used to identify optimal locations for long-term study sites and to implement management strategies.

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