# Chapter 11
# Climate Change Impacts on Soil Processes in Rangelands

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## 11.1 Introduction

Rangelands are terrestrial ecosystems dominated by unimproved vegetation communities that occupy about half of the land area of the globe, or about 67 million km$^2$ (WRI 1996). The term "rangelands" generally applies to areas with arid, semi-arid or dry sub-humid climates that are unsuitable for broad-scale farming (Harrington et al. 1984). Climatic variables, particularly rainfall and temperature, are the major drivers of ecosystem productivity and stability (and therefore of soil health) in rangelands because they directly influence soil moisture and nutrient pools. Areas receiving <250 mm annual rainfall are termed arid, while those with 250–500 mm are semi-arid. Rainfall is highly variable from year to year with a coefficient of variation of annual rainfall typically >30%.

Rangeland soils store more than 10% of terrestrial biomass carbon (C), and up to 30% of global soil organic carbon (SOC) (Schlesinger 1997; Scurlock and Hall 1998). While C sequestration rates are low in rangelands relative to those in environments that regularly support arable farming, their soils have an enormous potential to sequester C simply because of the vast area they occupy. Declines in

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plant production in rangelands resulting from lower precipitation directly affect soil organic matter (SOM) levels in the soil, given the dependence of SOC stocks on plant biomass (Dermer and Schuman 2007). In the Patagonian steppe, for example, simulations experiments indicate that effects of extreme drought on primary productivity can extend well into the following years, suggesting that droughts are likely to have an enduring effect on production (Yahdjian and Sala 2006).

Soil health in rangelands is intimately linked to vegetation cover and biomass through the production and storage of SOM. Managing healthy soils, therefore, involves managing vegetation cover, its biomass and composition. Thus, primary avenues for maximizing C retention and sequestration in rangelands are (1) manipulating grazing intensity through management of stocking rates, at various spatial scales, (2) revegetating areas of reduced native vegetation and (3) ameliorating ongoing erosion (Howden et al. 1991; McKeon et al. 1992; Walker and Steffen 1993; Conant and Paustian 2002; Henry et al. 2002). Many rangeland soils are unfortunately, heavily degraded (Ojima et al. 1993; FAO 2004), and there is even potential to lose C from otherwise, well-managed soils through continued grazing and frequent burning (see Sect. 11.5). Overgrazing or unsustainable farming of fertile patches of rangeland is likely to increase under increasing pressure from climate change.

The issues facing soil health in rangelands in the face of changing climate are extremely complex, being dependent on site history, the condition of the extant vegetation, the land management practices imposed and the dominating influence from a range of changed climatic attributes. There is also considerable uncertainty associated with the assessment of both the extant SOC stock, current greenhouse gas emissions and the potential sink in rangelands. Quantitative analyses based on actual measurements of SOC stocks and losses for many of the world's rangelands are rare (Conant and Paustian 2002).

In this chapter, we describe the main influences that a changing climate ~~is~~ are likely to have on rangelands ecosystems worldwide, with an emphasis on soils and soil processes. While a substantial amount of our knowledge is drawn from information for Australian landscapes, we use examples from rangelands worldwide to illustrate both the soil-level perturbations of changing climate and the more general ecosystem-wide impacts of a changing climate on both soils and vegetation.

## 11.2 Characteristics of Rangeland Ecosystems

Rangelands are managed for a variety of uses including pastoralism, mining, tourism, conservation, native cultures, military zones and, occasionally, cropping. Globally, they provide forage production for about three-quarters of the world's domestic livestock (Rangelands Australia 2008). Although traditionally used for pastoralism, there is a growing recognition of their importance for other uses such as conservation, hunting and the provision of ecosystem goods and services, recreation and aesthetics (Grice and Hodgkinson 2002).

A distinctive feature of rangelands worldwide is that productive, resource-rich soil occurs in patches. These "fertile patches" have moderate to high levels of moisture and nutrients, and support a larger proportion of plant diversity and productivity than the intervening resource-poor matrix (Stafford Smith and Morton 1990; Bestelmeyer et al. 2006). Feedback processes reinforce the intensity of these fertile patches, which also determine the distribution and abundance of soil biota. Small changes in soil moisture and fertility result in relatively large changes in soil biota (Whitford 2002). Thus, any changes in the amount and distribution rainfall or temperatures will have substantial effects on soils and their capacity to function.

Globally, many of the world's rangelands are severely overgrazed, invaded by exotic pests or adversely affected by inappropriate management such as frequent burning or cultivation along drainage lines. While low levels of grazing may increase the incorporation and decomposition of surface-resident SOM into the surface soil layers, overgrazing leads to potential losses in belowground SOC (Dermer and Schuman 2007). In wooded rangelands, higher grazing intensity (often accompanied by frequent, low-intensity fires) has been associated with decreased abundance of shrubs, less coarse woody debris and fewer trees with hollows (Eyre et al. 2010). Overgrazing, and vehicular usage is commonplace near drainage lines (areas of alluvial soils with higher net primary productivity), and near artificial watering points. Distance from water is a primary determinant of reduced biomass, and erosion is a secondary factor (Sparrow et al. 1997). With such surface disturbances, the soil nutrients and moisture become decoupled (Sparrow et al. 2003), with likely reductions in SOC. The degraded nature of rangelands and their reliance on a relatively small proportion of the landscape for production and diversity make them vulnerable to large-scale shifts in climate, particularly changes in the amount and relative distribution of rainfall.

## 11.3 Climate Change Forecasts for the World's Rangelands

There is growing evidence that the earth has begun to experience the effects of a changing climate. The area of land surface experiencing protracted periods of below-average rainfall has increased from 10 to 15% in the early 1970s to greater than 30% by early 2000 (Dai et al. 2004). Climate change is already affecting South American rangelands, with a mean warming of 1–4°C forecast for the next 70 years, particularly over the tropics (Yahdjian and Sala 2008). Over the last century, the north-western USA has warmed 0.5–1.5°C, and the temperature is projected to rise an additional 2–5°C by the end of the century. This is likely to increase the frequency and variability of droughts and floods (Chambers and Pellant 2008). Climate change predictions for rangelands in the south-western USA include an increase in average temperatures by 3–4°C by 2030, and an increase of 8–11°C by 2090 (Archer and Predick 2008). Climate models forecast a 1–2°C temperature rise in arid Central Asia by 2050, particularly in winter (Lioubimtseva and Cole 2006). For the Middle East, models forecast an overall temperature increase of 1.4°C by

2050 or 4°C by 2100. The largest change in precipitation is forecast for the Eastern Mediterranean, Turkey, Syria and the Caucasus, with a decline in precipitation due to decreased storm track activity (Evans 2009). Finally, climate change is expected to increase temperatures in India by 3–6°C, with reductions in rainfall of 5–25% by 2100, particularly during winter (Prabhakar and Shaw 2008).

These global trends are expected to increase the vulnerability of arid and semi-arid rangelands to drought and fire, and represent major challenges for managing vegetation and soils. More frequent and higher-intensity rainfall events in particular, exacerbated by greater drying of surface soils, are likely to induce higher rates of runoff, sediment removal and erosion, leading to feedback effects on nutrient and SOC loss. Higher diurnal surface temperatures globally will likely increase the frequency of hot days and warm nights, decrease the frequency of frosts, increase fire risk and result in a general pattern of drying (e.g. Zaitchik et al. 2007), particularly in the mid-latitudes.

Climate projections for Australia indicate an increase in drought frequency and severity (particularly in the productive grasslands and open woodlands of the southeast); rainfall intensity and the number of dry days will also increase continentally (Stokes et al. 2008). Rainfall seasonality is also forecast to change, with significant reductions in winter rainfall in the south. Forecasted change in annual rainfall for Australia varies across the continent. For example: (1) a decrease of 2–5% by 2030 over much of Western Australia, western South Australia and the south-western Northern Territory, even for a low emission scenario (IPCC 2000), (2) changes from −2 to +2% by 2030 for different regions of eastern Australia (IPCC 2000), and (3) a decrease of up to 40% for the southwest of the continent (CSIRO and BoM 2007). Rainfall is not the only climatic influence on SOC, with temperature change having a significant impact (Cowling and Shin 2006). The temperature for continental Australia is forecast to increase by 1–5°C over the next century (Williams et al. 2009).

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### *11.3.1 Influences on Soil Through Altered Plant Processes*

The short-term effects of climate change on rangelands will be to reduce plant growth rates (through reduced soil moisture), and therefore cover and biomass, altering litter production (including root litter and exudates) and therefore soil microbial communities (Table 11.1). As soil moisture is the principal driver of primary production in rangelands (Noy-Meir 1973), declines in vegetation cover will severely reduce the soil's capacity to resist erosion, further diminishing soil productivity. In higher productivity grasslands, a scenario of increased frequency of wildfire is likely (Table 11.1).

Altered climate will almost certainly be accompanied by changes in plant community structure through increases in $C_3$ shrubs at the expense of $C_4$ grasses, leading to woody thickening (see Sect. 11.3.3.1). In some areas, replacement of shrublands by annual grasslands has led to increased fire frequencies, changing

**Table 11.1** Summary of the effects of changing climate on terrestrial processes in rangelands and the short- and long-term effects on soils and soil processes

| | | Indirect effects | |
|---|---|---|---|
| Climate components | Direct effects | Short term | Long term |
| Lower rainfall | Increased drought severity, frequency and duration | Reduced plant cover | Lower inputs of soil organic matter |
| Altered rainfall distribution | Reduced soil moisture | Reduced plant biomass | Altered mineralization rates |
| Altered rainfall intensity | Increased rainfall erosivity | Altered litter fall | Reduced soil aggregation |
| Increased atmospheric $CO_2$ | Altered photosynthetic capacity | Altered microbial communities | Altered C:N ratios |
| | | Increased soil erosion | Increased soil bulk density |
| | | Increased frequency of wildfire | Altered spatial distribution of soil nutrients |
| | | | Reduced decomposition |
| | | | Reduced infiltration |
| | | | Increased coefficient of runoff |
| | | | Altered soil microbial communities |
| | | | Reduced termite populations |
| | | | Altered plant species composition |
| | | | Changes in ratio of $C_3$:$C_4$ plants |
| | | | Reduced habitat value |
| | | | Woody thickening attenuated by wildfire |
| | | | Exotic plant (weed) invasion |
| Increased atmospheric temperatures | Higher soil surface temperatures | Woody thickening | Altered cover of biological crusts |

these communities from a C sink to a C source (Chambers and Pellant 2008). The influences of these changes on ecosystems could be devastating, with changes affecting regional albedo levels, resulting in feedback effects on increased evapotranspiration, loss of soil moisture and ultimately rainfall decline (Chambers and Pellant 2008). There are also likely to be increased invasions of exotic species. For example, in Australia a potential threat to rangeland SOC is the introduced species buffel grass (*Cenchrus ciliaris*), which can retain high senescent biomass, increasing wildfire intensity and extent, and depleting native shrub biomass through multiple pathways (Butler and Fairfax 2003), and consequently also reducing SOC in the long term.

Changes in plant community structure and composition will have indirect flow-on effects to soil microbial communities, which respond to substrate chemistry (Waldrop and Firestone 2006), and this could compromise the ability of soils to retain C stocks. Conditions of high soil moisture, higher temperatures and grazing-induced disturbance could lead to a flush of microbial activity, depleting labile forms of soil C through microbial respiration (Killham 1994). This would lead to substantial reductions in soil aggregation and water-holding capacity, as well as cascading effects of increased runoff and sediment loss, further exacerbating the diminished capacity of the soil to sequester C and retain nutrients (Fig. 11.1). Soil health, climate and rangeland management are therefore intimately linked.

Changes in the C:N ratio of plant material would alter decomposition rates and the spatial distribution of soil nutrients. Perennial grasses have considerable capacity to store C as labile forms such as mucigels and polysaccharides on belowground tissue, which also supports a diverse soil microbial community (Whitford 2002). However, more labile C inherently has a faster turnover rate (i.e. lower longevity) than the C associated with the roots of woody plants (trees and shrubs), which can have different microbial associations with those of grasses. With its higher content of aliphatic suberin, waxes and lignin, the decomposition product of woody roots

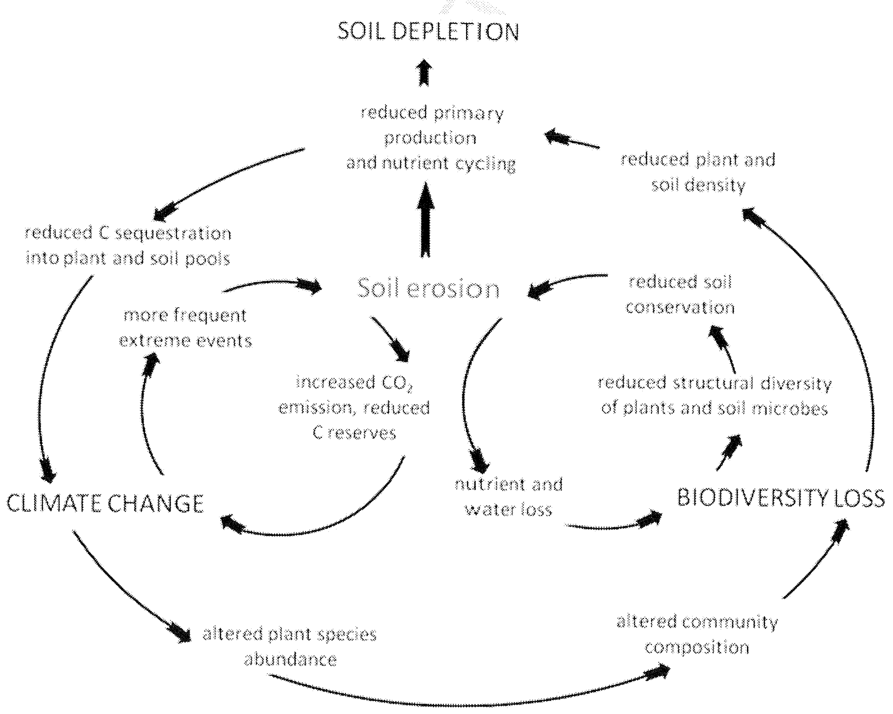

**Fig. 11.1** Interrelationships and feedbacks ~~between~~ climate change, biodiversity loss and soil health. Adapted from Millennium Ecosystem Assessment (2005)

among ✓

has a longer turnover time, thereby contributing slowly to SOC sequestration but forming a carbon pool of higher stability and hence a higher SOC stock in the long term.

### 11.3.2 Soil Feedbacks on Climate Change

Forecast changes in rainfall amount, distribution and intensity, increased temperatures and atmospheric $CO_2$ concentration will have substantial influence on: (1) retention and sequestration of C, (2) nutrient cycling, (3) resistance to erosion and (4) maintenance of functional hydrological processes (*sensu* Tongway 1995). Changes in rainfall will alter rangeland soil processes directly due to increased drought severity and soil surface temperatures, and exposure of the surface to radiation.

Accompanying changes in the rates and magnitudes of C fluxes between the atmosphere and vegetation will be changes in fluxes between vegetation and soil, with consequent flow-on effects to future climates. Positive climate change feedback may add 18% to atmospheric $CO_2$ levels by 2100 (Bonan 2008). Emissions from soil, reduced biomass and ecological impacts are part of this positive feedback cycle in climate change. The effects on soil processes are dependent on a large number of interacting factors including moisture, temperature, fire, human population demands and the degree of surface disturbance.

Rangeland management practices are also direct and immediate drivers in the positive feedback cycle, with deforestation (and prevention of regrowth) for cattle grazing a major source of greenhouse gas emissions, locally reduced rainfall and locally increased temperatures (and possibly higher drought severity) in Australia, Brazil and Columbia (McAlpine et al. 2009a, b). Fire is a major feedback mechanism wherever the evapotranspiration balance changes towards warmer and drier landscapes, while sufficient biomass remains to carry fire. This phenomenon applies to rangelands globally and to some higher productivity landscapes such as temperate forests.

Increasing temperature could alter the balance between influx and emission of C from the soil to produce a net emission, i.e. a net release of $CO_2$ back into the atmosphere. The activity of soil microbes is sensitive to changes in temperature and moisture. In areas such as Africa's Kalahari Desert, increases in temperatures could lead to a greater release of $CO_2$ to the atmosphere as C mineralization increases (Thomas et al. 2008). Rising temperatures will lead to significant increases in microbial respiration, resulting in an increased rate of SOM decomposition (Kirschbaum 1995). The effect could be to further exacerbate changing climate by accelerating the loss of SOC (Cox et al. 2000). Thus any increase in C sequestration resulting from the $CO_2$ fertilization of photosynthesis could be offset by increased microbial respiration (see Chap. 7).

### 11.3.3 *Climate-Induced Changes in Shrublands and Grasslands*

We consider now some of the interrelationships and feedbacks between climate, soils and rangeland management by examining two rangeland communities where climate-induced change is inevitable. Although Australian rangelands are unique in that they are predominantly shrubland and woodland with lesser amounts of scrub, heath and grassland, they share a common history of overgrazing effects with the Americas and southern Africa (Pickup 1998). We consider here community-level changes in desert shrublands in the western USA and the semi-arid grasslands of Australia, highlighting the importance of feedback mechanisms between altered climate, and altered vegetation and soil processes. These two case studies illustrate the close interconnections between climate, altered biodiversity and reduced ecosystem function, and therefore the effects of altered climate on rangeland soils (Fig. 11.1).

#### 11.3.3.1 Changing Climate Exacerbates Woody Thickening

The increase in numbers of woody plants, and encroachment of woody plants into grasslands (called "woody thickening"), is a global phenomenon (Archer et al. 1995). Dramatic increases in the density and extent of shrubs such as mesquite (*Prosopis glandulosa*) and creosote bush (*Larrea tridentata*) have occurred in the western USA (Buffington and Herbel 1965). Interactions among grazing, drought, rainfall events and reduced wildfire are thought to be significant drivers of woody thickening (Archer 1994; Grover and Musick 1990). Both climatic change and the more fundamental increase in atmospheric $CO_2$ concentration are thought to be responsible for woody thickening (Archer et al. 1995; Brown and Thorpe 2008).

Predictions of reduced rainfall and more frequent drought for south-western USA are likely to result in reduced cover of herbaceous desert vegetation (Brown and Thorpe 2008) and woody thickening of grasslands (Archer 1994). An increase in the frequency and severity of wildfires, a further consequence of climate change, would likely favour the proliferation of shrubs by removing competition for resources by grasses as well as reducing biological soil crusts. However, fire frequency or intensity above a critical threshold will have the reverse effect of reducing woody shrub density (Hodgkinson et al. 1984).

Woody thickening changes the scale at which soil nutrients are distributed, from a fine scale, which corresponds to the distribution of former grass tussocks, to a scale consistent with the average spacing of shrub hummocks (Schlesinger and Pilmanis 1998). Water, sediment, dust and airborne nutrients tend to accumulate under shrub canopies due to wind and water processes. Shrub-free (bare) interspaces experience higher surface temperatures and evapotranspiration, reduced organic matter incorporation, increased erosion and, on fine-textured soils where water accumulates, denitrification and ammonia volatilization (Schlesinger et al. 1990). Increased infiltration below shrub canopies can result in higher concentrations of tissue nitrogen

(N) (Bhark and Small 2003) through enhanced micro-arthropod activity and greater rates of decomposition. Well-developed tap roots allow semi-arid shrubs to access water from greater depths (Archer et al. 2002), thereby establishing microbial communities and SOC sequestration at depths greater than that of the former grasses.

Land-to-atmosphere feedbacks are also likely to occur at regional scales. Conversely, reduction in woody cover has been shown to reduce local rainfall through changes in heat flux, surface roughness, evapotranspiration and decreased cumulus cloud formation (McAlpine et al. 2009a), suggesting that a denser woody cover would increase local rainfall. These are conflicting scenarios, but the most torrid and drying climatic changes will precipitate a net loss of biomass. Due to the land degradation often associated with woody thickening, there is some belief that converting thickened areas back to grasslands would increase SOC. However, woody shrubs are effective at belowground C sequestration (Hibbard et al. 2003; Bai et al. 2009) and represent an opportunity to increase C stocks, even if animal production is adversely affected. Shrub removal, however, would likely induce net C emissions.

#### 11.3.3.2 Ecological Complexity Under Climate Change: The Semi-Arid Grasslands of South-Eastern Australia

Climate projections for the winter-dominant, semi-arid grasslands of south-eastern Australia indicate substantially less winter rainfall (80% decline) by 2070 (DECC 2008). Increased rainfall variability and frequency of high-intensity storms are likely to have substantial impacts on vegetation and soil. The effects most likely lead to replacement of relatively drought-intolerant bladder saltbush (*Atriplex vesicaria*) with an arid-adapted, drought-tolerant black bluebush (*Maireana pyramidata*) community. The cover and diversity of native grasses, herbs and forbs are also predicted to decline in response to greater soil moisture stress, and be replaced by arid-adapted Mediterranean weeds or grazing-tolerant forbs such as copperburrs (*Sclerolaena* spp.).

Reduced rates of C sequestration into aboveground and belowground pools will lead to reductions in abundance and diversity of aboveground biota, with feedback to the structure and diversity of belowground communities. Reductions in winter rainfall in these semi-arid grasslands will also increase the frequency of erosion events and the loss of soil nutrients. If grazing pressures are maintained, the risk of wind erosion of sandy soils will increase, particularly if plant cover drops below approximately 60% (Leys and Heinjus 1992). Reductions in winter rainfall, combined with greater levels of erosion, will also lead to a reduced cover of biological soil crusts (see Sect. 11.4) and further surface soil destabilization.

Soil and vegetation effects are likely to exacerbate changes in soil faunal populations, particularly termites, although we are unaware of empirical data for this area of grasslands. Research elsewhere in semi-arid grasslands indicates that termites are important components of healthy rangeland soils due to the range of ecosystem functions that they moderate. These functions are as broad as enhancing water flow into soils (Elkins et al. 1986), litter decomposition (Holt and Coventry 1988; Brown and Whitford 2003), C mineralization, nutrient recycling and subsequent plant production, particularly in low fertility soils (Parker et al. 1982; Coventry et al. 1988). Diminished grass cover will reduce abundance of termites, the main invertebrate decomposers in semi-arid and arid grasslands (Whitford 2002).

Termites are also preferred food items for a range of vertebrates and invertebrates. Replacement of grasses by exotic plants has potentially devastating bottom-up effects on semi-arid ecosystems by reducing termites, thereby reducing mineralization of N and C at landscape scales (Whitford 2002). This would appear to indicate a net reduction in C emissions with a reduced termite population. However, termites and other macro-arthropods also enhance water flow through soils by creating soil micropores (Eldridge 1994). Thus, reduced termite populations will reduce soil porosity and water storage (Whitford 2002) with consequent reduced water availability for plant growth.

## 11.4 The Potential of Rangeland Soils to Retain and Sequester Carbon

### 11.4.1 Plant Cover and the Maintenance of Healthy Soils

The most appropriate strategy for managing rangeland soils is to manage surface cover, either vascular plants (grasses, herbs, shrubs, trees) or non-vascular plants which make up the biological soil crust. Plant cover buffers the effects of wind and water on surface soils and therefore reduces the potential for erosion (Greene et al. 1994; McTainsh and Leys 1993). Below, we outline the importance of cover of both vascular plants and biological soil crusts for maintaining healthy rangeland soils, give some examples of how cover is likely to be affected by changing climate and describe the potential soil and ecosystem consequences of such changes.

Vascular plant cover has a major role in protecting soils against erosion, maintaining C stocks, and therefore improving soil health. Strategies to manage grazing in rangelands aim to manage plant cover and therefore maintain a range of critical physical, chemical and biological soil functions such as water-holding capacity, soil aggregation, surface stability and nutrient cycling. Plant cover also reduces raindrop impact and restricts the development of physically induced surface seals that impede water infiltration and prevent seedling emergence (Valentin and Bresson 1992). A major objective of rangeland management therefore is to maintain sufficient surface cover, which depends on soil type, rainfall amount and erosivity, soil moisture, slope and soil type (Greene et al. 1994). These critical cover thresholds for erosion prevention are thought to be in the range of about 40% (Greene et al 1994; Eldridge and Koen 2003).

### 11.4.2 Cyanobacterial Soil Crusts: Carbon Flux and Nitrogen Pools

Cyanobacteria are common components of soil crusts, along with lichens and mosses (Eldridge 2001a). Together these crusts stabilize the soil against water and wind erosion, regulate water flow into soils, provide a source of SOC and play vital roles in the maintenance and regulation of ecosystem functions. Cyanobacteria can survive soil temperatures of 50°C for prolonged periods and up to 100°C for 48 h (Rogers 1989). Cyanobacterial crusts can also sequester large volumes of $CO_2$, and in studies in the Mojave Desert, average net ecosystem exchange taken over a 2-year period ranged from 1 to 4 $\mu$mol-$CO_2$ $m^{-2}$ $s^{-2}$ net productivity. This is equivalent to about 1 t C $ha^{-1}$ $year^{-1}$ (Wohlfahrt et al. 2008). Cyanobacterial crusts from soil and rock at sites in western Queensland sequester 0.5–1.8 $\mu$mol-$CO_2$ $m^{-2}$ $s^{-2}$ (Wendy Williams, personal communication, 2009). These crusts therefore have the capacity to sequester up to 1 t C $ha^{-1}$ $year^{-1}$ (~1.7 million t C $year^{-1}$ for Australian rangelands).

Increases in dust storm frequency resulting from changing rainfall patterns (McTainsh and Lynch 1996) may alter the ability of cyanobacterial soil crusts to produce N. Cyanobacteria and cyanolichens fix substantial quantities of N in rangeland soils (Smith et al. 1990). However, increased frequency of sand storms may reduce these quantities. In western Queensland, landscape-scale deposition of coarse sand in the semi-arid woodlands is associated with an increase in soil N pools (Williams and Eldridge, unpublished data). Sand deposition leads to autolysis of N-enriched cyanobacterial cell material and therefore greater soil N pools. While stored N gradually accumulates in surface soils, long-term N production from these soils is compromised, reducing surface soil stability. Any inappropriate land management that leads to an increase in sand deposition (e.g. overgrazing) is likely to lead to long-term reductions in soil N pools.

### 11.4.3 Climate Change Impacts on Lichen-Dominant Crusts

Lichens cannot tolerate high summer temperatures combined with high humidity or rainfall because it reduces the photosynthetic ability of the algae component. In Australia, this intolerance to summer rainfall limits the distribution of lichen crusts to mainly winter-dominant areas in the south. Thus, while lichen crusts can tolerate surface temperatures in excess of 70°C, a temperature of 30°C for 30 min when fully hydrated is fatal (Rogers 1989). Under current climate change forecasts for Australia, lichen crusts are likely to be lost over significant areas of semi-arid southern Australia as summer rainfall is likely to increase in some areas that are currently winter dominant. The effect of lichen reduction may be catastrophic given their pivotal role in soil stability (Eldridge 2001b), and may lead to reduced landscape stability, with varied effects on water flow and wind erosion.

Compositional shifts in soil crusts, however, may mitigate against ecosystem collapse because cyanobacteria are likely to dominate the crusts as summer rainfall increases. It is possible, therefore, that cyanobacteria will play a moderating role in soil stabilization in the face of long-term climate change (Rogers 1989).

## 11.5 Grazing and Burning Exacerbate the Effects of Climate Change on Rangeland Soils

### *11.5.1 Grazing Effects on Rangeland Soils*

Any negative effects of a changing climate are likely to be exacerbated by overgrazing, as stocking rate is a major driver of vegetation and soil change in rangelands (McKeon et al. 2009). Overgrazing leads to a range of soil-related problems including compaction (Thurow et al. 1988), soil fertility and nutrient decline (Snyman 1999), and loss of structural integrity (Thurow et al. 1988). Given projections of lower rainfall and higher temperatures in rangelands globally, continued grazing will likely alter the structure or plant communities, with potentially irreversible effects on ecosystem resilience.

Although grazing is a useful tool for managing vegetation and therefore influencing decomposition and soil nutrient levels, its effects on soils are site-specific (Beukes and Cowling 2003). Some grazing practices such as low-risk, opportunistic grazing may have little effect on soils. However, there are likely to be substantial soil effects under continuous grazing. Tactical grazing strategies, whereby larger numbers of animals graze smaller paddocks over shorter time periods, are thought to provide ecological benefits to the soil in some rangelands (e.g. South African grassland) where rainfall is more reliable. However, in many seasonally variable rangelands such as the semi-arid woodlands of eastern Australia or the grasslands and woodlands of the USA, trampling associated with rotational grazing has substantial adverse effects on soils such as destruction of the biological soil crust and reduction in infiltration rate (e.g. Weltz et al. 1989).

### *11.5.2 Savannah Burning in Subtropical Australian Rangelands*

Australia's northern savannas can be a net source or sink of C depending on how they are managed. Under a regime of low-intensity biennial burning, savannahs are net C sequestrators (0.5–2.0 t C $ha^{-1}$ $year^{-1}$), with major contributions including emissions from fires (1.6 t C $ha^{-1}$ $year^{-1}$), sequestration due to tree growth (1.2 t C $ha^{-1}$ $year^{-1}$) and sequestration from woody thickening (0.2 t C $ha^{-1}$ $year^{-1}$) (Beringer et al. 2007). The present state of annual savannah burning, however, being a mixture of intense and moderate burns, contributes substantially to the national greenhouse gas budget (Williams et al. 2005). Similarly, high-intensity, late-season burns constitute an ongoing C efflux (Dyer and Stafford Smith 2003), which is often exacerbated by the consumption of large amounts of standing dead timber (Fensham 2005). This is accompanied by reductions in SOC levels and increased atmospheric aerosol concentrations. Modelling suggests that increased grass growth, a consequence of atmospheric $CO_2$ enrichment, coupled with increased wildfire, presents significant threats to net C sequestration in the semi-arid woodlands (Howden et al. 2001).

Careful management of livestock and grassy-fuel loads ~~are~~ necessary to maintain long-term sequestration rates above the present 1 t-C $ha^{-1}$ $year^{-1}$ (Williams et al. 2004) in the tropical savannas and to prevent this sink becoming a C source. Increased grass density may favour higher levels of grazing, ultimately promoting woody thickening, and converting open savannah into woodland (but possibly with initial erosion before SOC recovery through thickening). In Australia's semi-arid mulga woodlands, modelling indicates that century-long net sequestration and maintenance of aboveground C is only possible under a scenario of no fire and no grazing (Howden et al. 2001).

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## 11.6 Rehabilitation of Rangeland Soils

The vastness, degraded state and relatively low human population of the world's rangelands have prompted numerous studies of the effects of management changes on C sequestration (e.g. McKeon et al. 1992; Walker and Steffen 1993; Garnaut 2008). Howden et al. (1991) estimated that rehabilitating the rangelands globally to their pre-degraded state would sequester about 7 t-C $ha^{-1}$ as SOC, i.e. an extra 0.5 wt% organic C. Rangeland rehabilitation has long been considered uneconomical due to the low returns and long timelines involved (e.g. Perry 1974). It can, however, be profitable under soil C trading (e.g. Cerri et al. 2003). Also, woody thickening offers C sequestration even without investment in management intervention. Currently, projects assessing C sequestration in rangelands, while simultaneously improving land condition, are underway worldwide.

There are currently an estimated 4.5 Gt of SOC in the 380 Mha of Australia's commercially grazed rangelands to 0.3 m depth (with possibly another 40% to 2 m depth). Ongoing emissions from the soil due to management for grazing are ~5.3 Mt-C $year^{-1}$ (Dean et al. 2009). Alternatively, an increase in SOC by only 1% over those rangelands would equate to 45 Mt-C. However, the period for that sequestration is in the order of two centuries (Dean et al. 2009), owing to the slow rate of SOC sequestration in semi-arid rangelands (Hibbard et al. 2003; Singh et al. 2007). When sequestration in biomass and coarse woody debris are included (from both woodland regrowth and the slower process of land rehabilitation), the sequestration is much faster and estimates for Australia range from 0.273 to 3.2 t-C $ha^{-1}$ $year^{-1}$ over periods of 20–140 years (Moore et al. 2001; Garnaut 2008;

Dean et al. 2009). Rehabilitation of Australian mulga-lands has been forecast to sequester up to 68 Mt-C $\text{year}^{-1}$ over several decades (Garnaut 2008). At the national scale, however, the current trend of a net emission from rangeland soils must be slowed and reversed before sequestration is manifested.

## 11.7 Rangeland Soils Under a Changing Climate: Concluding Remarks

Climate change will intensify problems currently facing rangeland managers worldwide (FAO 2004; McKeon et al. 2009). Declining pasture productivity, reduced forage quality, increased livestock heat stress, more frequent weed and pest invasions, more frequent droughts, less frequent, but more intense rainfall and greater soil erosion are all likely outcomes of the projected changes in climate (Stokes et al. 2008; McKeon et al. 2009). Although the magnitude of these effects and their feedbacks are not well understood (Henry et al. 2007), they will have major implications for rangeland managers. How rangeland managers adjust to these different impacts will, in turn, have major feedback effects on the extent of climate change.

Climate change is expected to increase the vulnerability of arid and semi-arid rangelands to further degradation. With climate change effects already evident in the rangelands, and with an increasing human population, both mitigation and adaptation (to climate change) are imperative. Several avenues for mitigation in the rangelands have already been broached (Henry et al. 2002; WAG 2002; FAO 2004; Foran 2007; Dean et al. 2009; Dwyer et al. 2009; Fensham and Guymer 2009; McAlpine et al. 2009b). They include (1) protection and enhancement of SOC through, for example, management of grazing intensity, (2) reduced clearance of native vegetation, (3) regrowth of areas previously deforested for grazing, (4) planting of deeper-rooting vegetation, (5) increasing scientific knowledge of physical and ecological processes and revision of institutional frameworks for knowledge integration, (6) control of invasive plants and animals, and (7) implementation of policy initiatives aimed at C sequestration. Although requiring gross changes to current practices, these avenues can be simply distilled as constituting careful management of soil and ecological processes, biodiversity and vegetation cover. Adapting to climate change principally involves a concerted application of resources and management effort to these very same issues. Accordingly, adaptation will require action at a governmental level, establishing national and international funding to the avenues listed above, along with integrated management towards these activities (e.g. FAO 2004).

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