

Climate change impacts on wind and water erosion on US rangelands

B.L. Edwards, N.P. Webb, D.P. Brown, E. Elias, D.E. Peck, F.B. Pierson, C.J. Williams, and J.E. Herrick

Abstract: Soil erosion by water and wind in US rangelands has serious implications for rangeland health and food security and poses significant hazards to human health and communities. Accordingly, understanding how future climate change may impact soil erosion is critical for developing appropriate management strategies that mitigate negative impacts to the extent practical and potentially build resilience. Here, we review potential impacts of climate change on controls of erosion in US rangelands and discuss potential erosion responses. Projected climate changes are expected to have regionally variable effects on important controls of erosion, especially vegetation cover; community composition; frequency, magnitude, and geographical range of fire disturbance; and high intensity, erosive weather events, all of which have the potential to increase rangeland vulnerability to erosion. We identify knowledge gaps relevant to these controls and discuss management considerations to address climate change impacts to soil erosion concerns for US rangelands. In order to improve resilience and the efficacy of climate change adaptation, we recommend that existing monitoring data be used to create assessments of vulnerability, that soil erosion should be explicitly included in management benchmarks and decision support tools, and that no-regrets management options be implemented in anticipation of future impacts.

Key words: climate change—rangeland—soil erosion—water erosion—wind erosion

Understanding the potential impacts of projected climate change on soil erosion in US rangelands is critical for developing forward-looking, sustainable land management practices for these high-value ecosystems. Rangelands cover approximately ~31% of total US land area (Havstad et al. 2009)—about 60% of western states (figure 1)—and provide a critical resource for food production in addition to other high-value ecosystem services (Mitchell 2000). Criteria for delineating rangelands vary depending on source (Lund 2007; Reeves and Mitchell 2011), but in general they can be defined as arid and semiarid systems with low primary production because of highly variable precipitation (Havstad et al. 2009; Briske et al. 2015). Rangelands are largely vegetated by grasses, forbs, and shrubs, but include diverse subenvironments distributed over the western United States, each with unique ecologies and management concerns (figure 1). Because they exist near physiological thresholds, rangelands are highly sensitive to climatic stressors; consequently, rangelands

and the communities and industries they support are particularly vulnerable to climate change (Archer and Predick 2008).

Predicting how rangelands will respond to climate change is difficult because of often large uncertainties in climate projections (Wuebbles et al. 2017), spatial variability in responses across vegetation communities and soil types (Webb et al. 2012; Briske et al. 2015), and ecogeomorphic feedbacks that may produce threshold-type responses (Archer and Predick 2008; Okin et al. 2018). Still, projected climate trends provide a useful framework for identifying potential impacts and highlighting knowledge gaps, research foci, and management considerations. Several reviews have discussed critical impacts of climate change on rangelands, including assessments for plant community responses (McKeon et al. 2009; Webb et al. 2012; Polley et al. 2013; Briske et al. 2015), ecosystem services (Palmquist et al. 2016; Boone et al. 2018), and livestock production systems (Howden et al. 2008; McKeon et al. 2009), as well as socioeconomic impacts and adap-

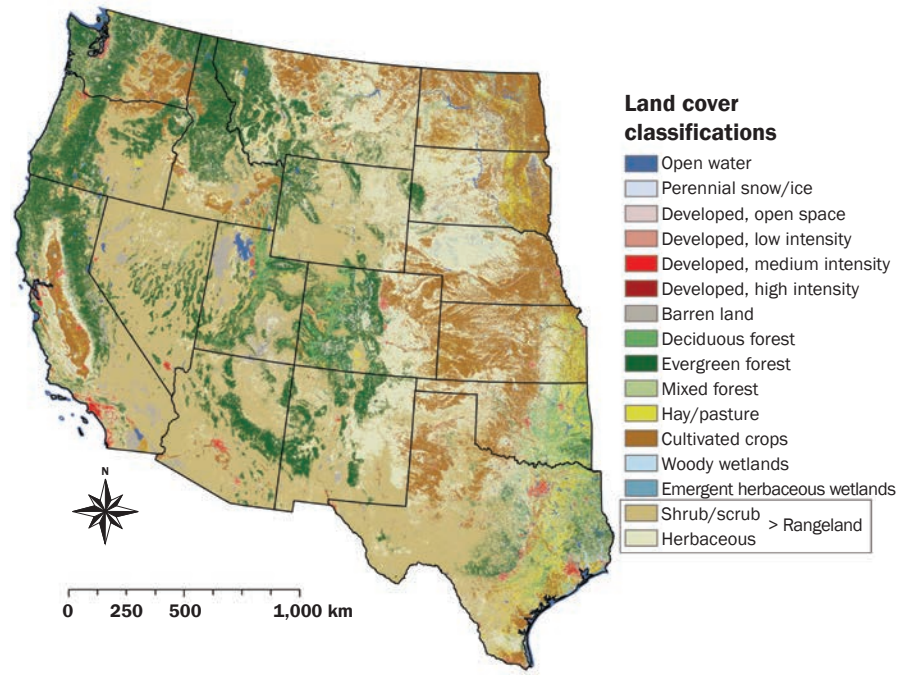
tation strategies (Thornton et al. 2009; Joyce et al. 2013; Webb et al. 2013). Fewer papers, however, (Nearing et al. 2005; Munson et al. 2011; Zhang et al. 2012; Pu and Ginoux 2017) have focused specifically on how climate change will impact key geomorphic processes such as soil erosion, and these are largely regional- or process-specific.

A significant portion of US rangelands are already vulnerable to soil erosion (Weltz et al. 2014). Soil erosion by water and wind is an important agent of land degradation that impacts nutrient availability, forage production and food security, and poses serious hazards to human health and communities. Conceptually, soil erosion on rangelands can be framed as a climate- and land management-driven balance between (1) largely biotic controls that resist sediment detachment and transport, e.g., vegetation, ground cover, and soil organic matter; and (2) abiotic processes that drive erosion, e.g., rainfall, overland flow, wind, and disturbance (Pierson et al. 2011; Turnbull et al. 2012; Williams et al. 2014b, 2016b). Rangelands are often sparsely vegetated, and the amount, type, and distribution of vegetation and ground cover strongly control infiltration, soil retention, and exposure of soils to potentially erosive rainfall and winds (Okin et al. 2006, 2018; Webb et al. 2014; Pierson and Williams 2016). Runoff, waterborne soil loss, and wind-driven sediment transport are typically low on well-vegetated sites. Cover reductions associated with land use- and/or climate-driven disturbances commonly result in amplified runoff and water erosion and greatly increase potential for aeolian transport and dust emission (Turnbull et al.

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Figure 1

Land cover classifications of the western United States based on the National Land Cover Database (2011). Rangeland, represented here as shrub/scrub and herbaceous land cover, represents 55.22% of the total land area of the western contiguous United States. Source: Homer et al. (2015).



2008; Pierson et al. 2011; Field et al. 2012; Turnbull et al. 2012; Williams et al. 2014b; Okin et al. 2018; Webb and Pierre 2018). In addition, disturbances such as wildfire and infrastructure development (e.g., for oil and gas extraction) can increase risks of runoff, flooding, and water and wind erosion events during recovery periods, which can be prolonged by drought (Sankey et al. 2009; Wagenbrenner et al. 2013; Miller et al. 2012; Pierson and Williams 2016). Further, plant community transitions that increase fire activity pose an even greater risk for long-term soil loss through frequent repeated burning (Pierson et al. 2011; Wilcox et al. 2012; Williams et al. 2014b, 2016c).

Predicting climate change impacts on rangeland soil erosion and identifying appropriate management strategies requires understanding the effects of climate on both resisting and driving controls. This paper provides a broad overview of the potential impacts of climate change on these controls for US rangelands. We then highlight potential water and wind erosion responses and identify key knowledge gaps regarding climate change-driven increases in erosion. Finally, we discuss current and future management considerations.

Climatic Setting and Observed and Projected Climate Changes

US rangelands include portions of the Great Plains, Intermountain West, Great Basin, and Desert Southwest. These regions are most commonly characterized by arid to semiarid climates, but precipitation and temperature vary across season, elevation, and geography. Climatic variations and extremes are common in these areas and can include multiyear droughts, large wildfires, floods, convective storms, and the passage of frontal systems that produce regional dust storms (Brazel and Nickling 1986; Herring et al. 2018).

There is very high confidence that temperatures across most US rangelands have been warming over the past century, a trend that has intensified in recent decades (Vose et al. 2017). (Wuebbles et al. [2017] define “very high confidence” as strong evidence and high consensus among projections. “High confidence” is defined as having moderate evidence and medium consensus. “Medium confidence” is defined as having suggestive evidence but competing schools of thought. “Low confidence” is defined as having inconclusive evidence and disagreement among experts.) Indeed, some of the most pronounced increases in both winter and summer temperatures over the past 30 years

have occurred over the western half of the country (Vose et al. 2017). A lengthening of the frost-free season by 7 to 17 d y^{-1} has also been observed (Hibbard et al. 2017). Output from the Coupled Model Intercomparison Project (CMIP5) suggests that mean temperatures across the West could increase by $\sim 3.3^{\circ}\text{C}$ (6°F) by the mid-21st century, and $\sim 5^{\circ}\text{C}$ (9°F) by late-21st century (figure 2) under the Representative Concentration Pathway 8.5 (RCP8.5) (Vose et al. 2017), which is consistent with observed increases in global carbon (C) emissions over the past 15 to 20 years (very high confidence) (Wuebbles et al. 2017). Under this scenario, widespread increases in temperature are projected across all seasons, especially summer and fall (figures 2 and 3). Such increases could affect soil erosion processes through impacts on evaporative stress and soil moisture deficit (medium confidence), growing season length (low confidence), and plant community structure (high confidence) (Blumenthal et al. 2016; Hibbard et al. 2017; Wehner et al. 2017).

Observed changes in annual precipitation have been more varied across US rangelands, with drying in much of the Southwest and Intermountain West and wetter conditions across much of the Great Plains (medium confidence) (Easterling et al. 2017). Patterns are even more heterogeneous at seasonal time scales, with drying in the Southwest observed during spring and summer and drying in the Northwest and Intermountain West observed during winter (see Easterling et al. [2017] for detailed maps). It should be noted that projections for annual precipitation by the mid-21st century under RCP8.5 are more uncertain than those for temperature but suggest significant drying in the Southern Plains and Southwest and slight increases in precipitation over portions of the Northern Plains and Northwest (figure 2) (Greene and Seager 2016). Midcentury projections under RCP8.5 for seasonal precipitation suggest that much of the Southwest will have drier winters and springs but wetter summers; much of the Southern Plains will have drier springs and summers with drying across all seasons in southeast Texas; the Northern Plains will have wetter springs and drier summers; and the Northwest, while more spatially variable than other regions, will have generally wetter winters with some drying in fall and spring (figure 3) (Swain and Hayhoe 2015). Other observed trends

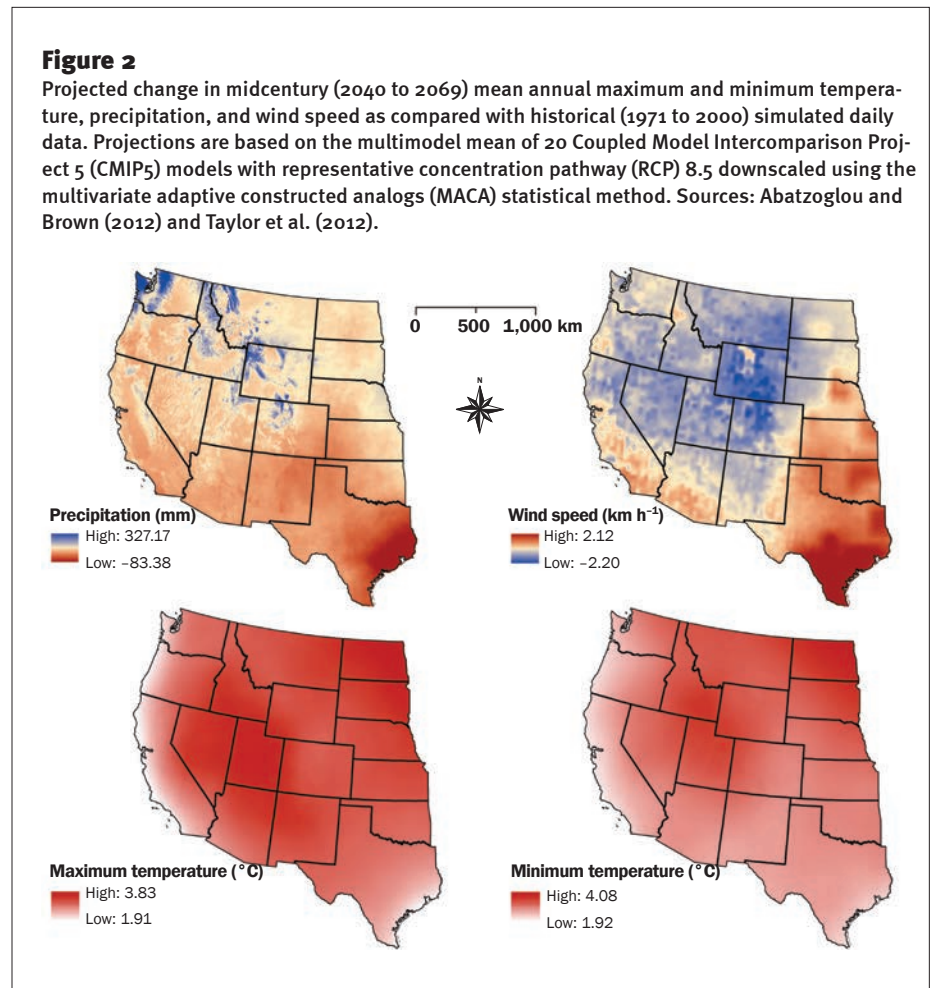
that are projected to continue through mid-century include increased frequency and intensity of heavy rainfall events (figure 4; high confidence) and a higher proportion of precipitation falling as rain instead of snow (high confidence) (Easterling et al. 2017). Climate change is also thought to be affecting the frequency, intensity, and location of extreme events that may impact erosion, such as large fires (medium confidence), severe thunderstorms (low confidence), and severe winter storms (low confidence) (Kossin et al. 2017; Wehner et al. 2017).

Surface wind speeds, beyond those during severe thunderstorms, were not addressed in detail by Wuebbles et al. (2017). However, Vautard et al. (2010) report that surface winds have declined across the northern midlatitudes over the past 30 years, a “stalling” of 10% in the United States. Large-scale circulations explain a portion of these changes, but so too may increased surface roughness due to land use change and land cover change (Vautard et al. 2010; Cowie et al. 2013). Patterns of seasonal wind speed projections for the mid-century are consistent with these evaluations (figure 3). Much uncertainty remains, however, about the complex physical connections between climate change, wind speed, and other extreme weather events (Seneviratne et al. 2012). What is certain is that wind and water erosion responses to climate change will be variable across US rangelands. Figure 4 summarizes key observed and mid-century climate change projections that may be particularly impactful for future trends in erosion.

Potential Impacts of Climate Change on Controls of Erosion

The potential for climate change to impact soil erosion from rangelands depends on its broad-scale effects on resisting and driving controls that define site susceptibility and vulnerability to erosion. At the site level, surface conditions that comprise resisting controls dictate susceptibility of the ground surface to erosion. Overall vulnerability to erosion is a function of surface susceptibility and exposure to driving forces, i.e., wind and water erosivity (figure 5) (Pierson et al. 2011; Williams et al. 2014b, 2016b).

Surface susceptibility is defined by the amount, type, and distribution of vegetation and ground cover (biotic structure), inherent soil properties (e.g., bulk density, erodibility, texture, and water repellency), surface roughness (e.g., vegetation height and spac-



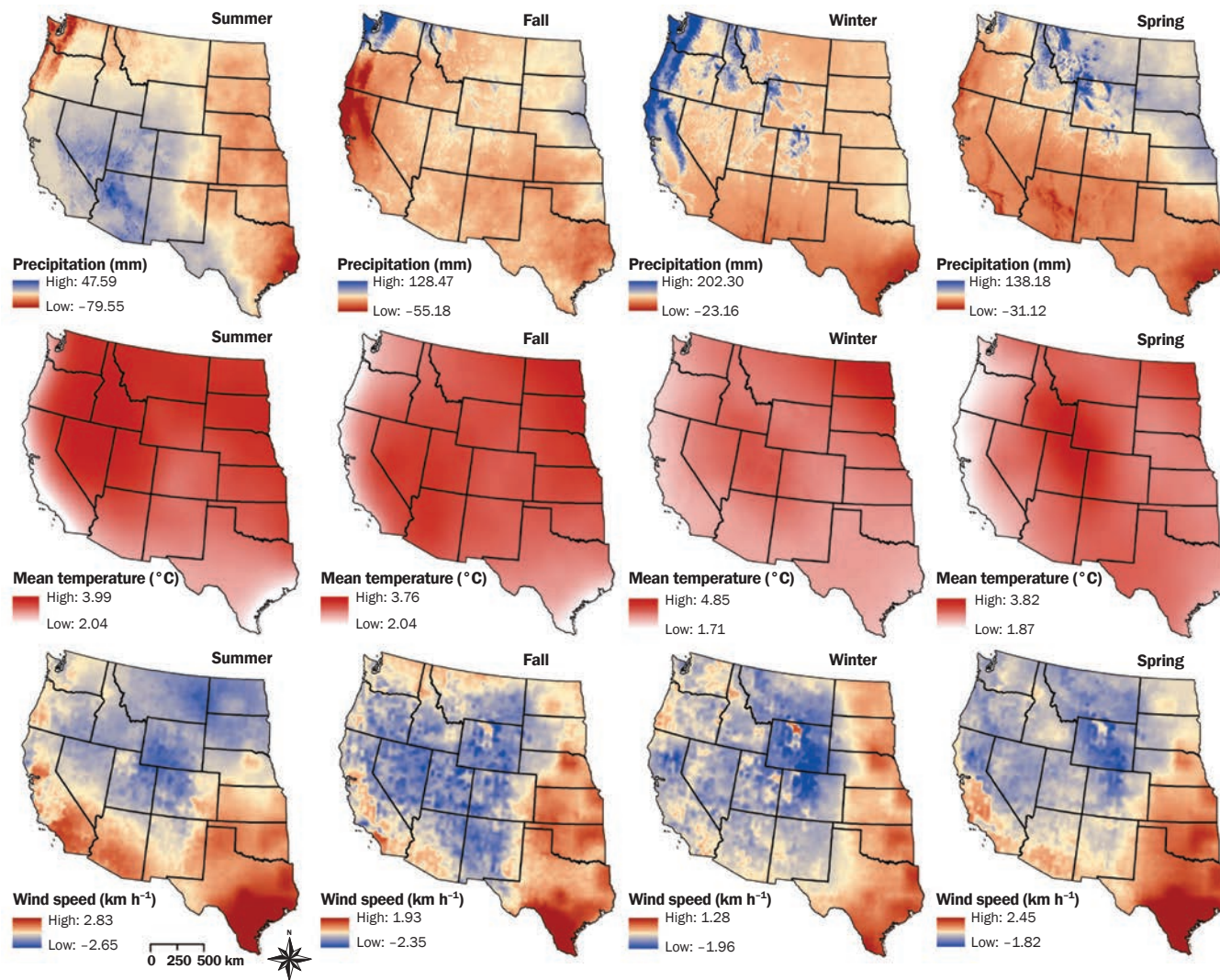
ing), and topography/slope steepness (Wolfe and Nickling 1993; Webb and Strong 2011; Williams et al. 2014b). Spatial heterogeneity in surface susceptibility and precipitation across a site create heterogeneous infiltration, runoff, and water erosion patterns that dictate losses or retention of water and sediment resources (Wilcox et al. 2003; Ludwig et al. 2005; Puigdefàbregas 2005; Pierson and Williams 2016). Sparsely vegetated or bare soil locations (source areas) exhibit high evaporative losses and low soil water storage (Huxman et al. 2005; Newman et al. 2010), promote runoff (Pierson et al. 2010; Turnbull et al. 2010; Williams et al. 2014b), and facilitate transfer of water and soil resources to areas with ample surface protection (sink areas) (Reid et al. 1999; Wilcox et al. 2003; Ludwig et al. 2005; Field et al. 2012; Pierson and Williams 2016). Vegetation, litter, and ground cover dampen the erosive energy of rainfall and overland flow and delay and reduce runoff and erosion by trapping water input, stabilizing sediment, and promoting infiltration (Webb et al. 2014; Pierson and Williams 2016). These same factors reduce

wind erosion by sheltering soil from the wind and attenuating shear stress imparted to the surface by the wind field (Wolfe and Nickling 1993). However, while fractional cover of vegetation, rocks, and litter is of primary importance to susceptibility to water erosion, wind erosion is a lateral process. Thus, lateral cover of plants (and other nonerodible elements), their structure, height, and spatial distribution on the landscape control surface aerodynamics (Chappell et al. 2018).

Accumulation of soil water and nutrients in sink areas stimulates belowground biological activity, plant growth, and reproduction that further sustain the vegetative community and the overall source-sink structure (Schlesinger et al. 1990, 1996; Ludwig et al. 1997; Wilcox et al. 2003; Belnap et al. 2005; Ludwig et al. 2005; Turnbull et al. 2012). Water and wind-driven sediment transport may also be complementary, with water-driven transport of soil and nutrients into eroded intercanopy gaps increasing the availability of sediment and nutrients for entrainment and transport by wind (Bullard and McTainsh 2003; Okin et al. 2006).

Figure 3

Projected seasonal change in midcentury (2040 to 2069) temperature, precipitation, and wind speed as compared with historical (1950 to 2005) simulated daily data. Projections are based on the multimodel mean of 20 Coupled Model Intercomparison Project 5 (CMIP5) models representative concentration pathway 8.5 downscaled using the multivariate adaptive constructed analogs (MACA) statistical method. Sources: Abatzoglou and Brown (2012) and Taylor et al. (2012).



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Journal of Soil and Water Conservation 74(4):405-418 www.sowcs.org

On well-vegetated and stable hillslopes, a feedback exists where vegetation and ground cover trap water and soil resources and promote soil properties that facilitate infiltration. Soil resource accumulation stimulates biological productivity that further sustains vegetation and ground cover, low surface susceptibility, and long-term ecological resilience. In the absence of disturbance, vulnerability to erosion and overall ecological resilience are therefore a function of the key characteristics that define an ecological site: (1) climate, (2) soil attributes, (3) plant community dynamics, and (4) landscape position (Williams et al. 2016b; Galloza et al. 2018). Alteration of vegetation structure

through plant community shifts, fire, or other disturbances can increase the susceptibility of the ground surface to runoff and sediment transport, particularly under intense rainfall or wind events (figure 5) (Davenport et al. 1998; Turnbull et al. 2008; Bergametti and Gillette 2010; Pierson et al. 2011; Wilcox et al. 2012; Webb et al. 2014; Williams et al. 2014b, 2014a).

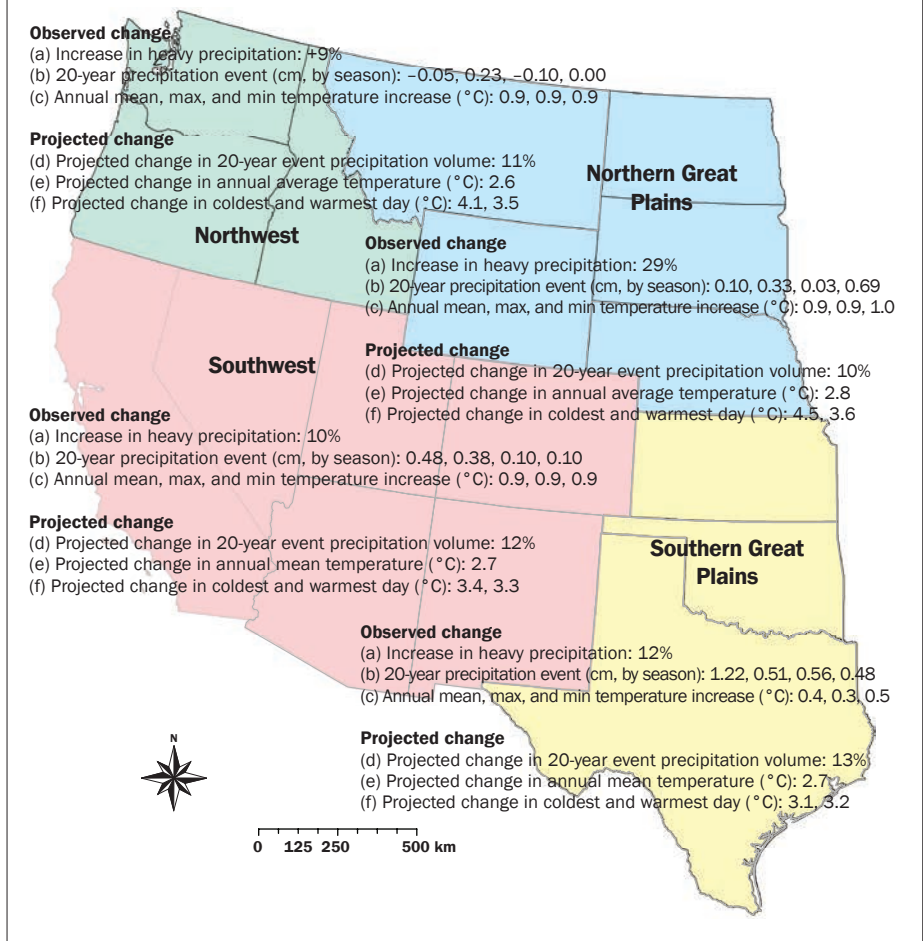
Whether climate change will have broad-scale negative impacts on soil erosion in US rangelands depends on (1) the degree to which it regionally alters controls of erosion that increase site susceptibility through altered disturbance regimes and plant community dynamics, and (2) how much it

increases overall vulnerability by altering trends in both surface susceptibility and driving forces, such as the amount, timing, and erosive energy of precipitation and wind. The following sections summarize the potential impacts of climate change on the dominant controls of erosion in rangelands.

Vegetation Response to Projected Climate Change. Projected changes in atmospheric carbon dioxide (CO_2) concentration, temperature, and precipitation will likely have significant impacts on vegetation production, cover, and community structure in US rangelands (Polley et al. 2013; Briske et al. 2015). Rangeland net primary production depends primarily on amount and timing of precip-

Figure 4

Regional observed and projected changes in temperature and precipitation. (a) Observed change in heavy precipitation (amount of precipitation that exceeds 99th percentile; 1958 to 2016). (b) Observed change in the daily 20-year precipitation event (cm; seasonal maximum precipitation totals: 1948 to 2015). (c) Observed regional increase in annual mean temperature, 1986 to 2016 to 1901 to 1960. (d) Projected percentage increase in daily 20-year precipitation event for mid-century scenario (RCP8.5). (e) Projected change in annual average temperature. (f) Projected temperature change for coldest and warmest day. Source: Wuebbles et al. (2017).



itation (Polley et al. 2013), while warming and increasing CO₂ concentrations contribute primarily to plant water use efficiency, evaporative demand, and growing season length (Polley et al. 2013; Briske et al. 2015). Vegetation community structure depends on competition for limited water resources and nutrients (Gherardi and Sala 2015), but also on abiotic feedbacks associated with wind and water redistribution of soil, soil nutrients, and C, as well as land use and management (Hibbard et al. 2017).

Increased CO₂ concentrations promote growth and allow for more efficient water use by plants, which has the potential to offset the effects of warming to some degree (Polley et al. 2013). However, these benefits will likely be limited by water availability in the Southern

Plains and Southwest. Polley et al. (2013) and Briske et al. (2015) suggest that coupled warming and drying trends in the Southwest and Southern Plains will reduce overall net primary productivity and could favor shifts in community composition to more woody species in the Southern Plains and Southwest. In contrast, increased CO₂ and warming coupled with wetter conditions in the Northern Plains will likely lead to increased productivity but may favor invasive C3 herbaceous species. Both agree that predicting vegetation response in the Northwest—where overall precipitation amounts are projected to be relatively stable with a seasonal shift to wetter winters and drier summers—is more uncertain, but that warmer, drier summers should reduce late summer productivity and

wetter, warmer winters could favor invasive annual grasses, particularly cheatgrass (*Bromus tectorum* L.), which has serious implications for increased fire activity (Balch et al. 2013; Pilliod et al. 2017).

Field studies have shown that increased interannual variability in precipitation decreases overall ecosystem productivity and that this effect becomes more pronounced over time (Sala et al. 2012; Gherardi and Sala 2015). Moreover, Gherardi and Sala (2015) demonstrated that shrub productivity in southwestern deserts increases in response to increased precipitation variability while grass productivity decreases, which suggests that prolonged periods of increased variability in precipitation could promote community transitions from grass-dominated to shrub-dominated landscapes. Shrub encroachment, once initiated, is often self-sustaining. High infiltration rates, deeper soil water storage, and entrapment of nutrient rich soils underneath or adjacent to shrub canopies create favorable conditions for shrub growth, often at the expense of herbaceous species (Schlesinger et al. 1990; Parsons et al. 1992; Bhark and Small 2003; Okin et al. 2009; Turnbull et al. 2010; Field et al. 2012). Overall, community transitions might occur, which favor erodibility and increase site susceptibility. Such transitions should be closely monitored for their effects on soil erosion.

Wind Erosivity. Mean wind speeds are projected to be lower for much of the western United States, except for the Southern Plains and portions of the Northern Plains regions (figure 2), but these projections are highly uncertain (Vautard et al. 2010). Regardless, this small reduction in potential erosivity will likely be offset in many locations by vegetation responses to climate change. Wind erosivity depends on how the wind field interacts with exposed soil at the surface, which is controlled by vegetation cover and structure (Mayaud and Webb 2017). Vegetation both extracts momentum from the wind and provides shelter zones in which wind speeds at the surface are reduced depending on plant density and size (Wolfe and Nickling 1993). Thus, trends in vegetation that promote sparseness or transition from dense-cover grasses to shrubs with exposed gaps can effectively increase wind erosivity (Munson et al. 2011; Okin et al. 2018). Further, while mean wind speeds are projected to be lower for most of the West, large-scale wind erosion and dust emission

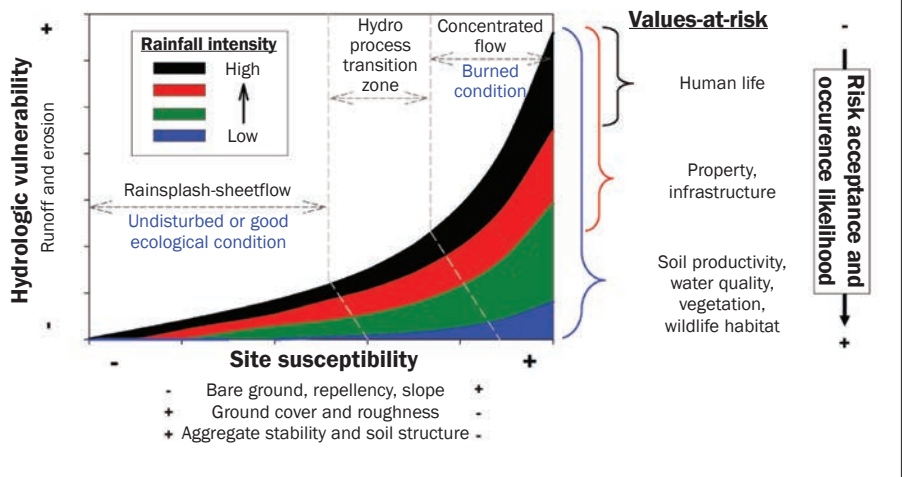
events are largely driven by frontal passages over much of the region. Drier winters and springs in the Southern Plains and Southwest may leave many sites more vulnerable to these erosive events by reducing early season production and affecting green-up timing, and increased fire activity in the Northwest could lead to more exposed surface area during recovery periods.

In addition to synoptic scale frontal wind patterns, local convective winds are also important drivers of wind erosion in US rangelands. Local dust events are often coupled with convective storms (Brazel and Nickling 1986; Novlan et al. 2007), which can account for a significant portion of dust activity/hazards. In general, climate models predict an increase in extreme thunderstorm events (Kossin et al. 2017), but confidence in these projections is low. Thus, relating increases in convective storm frequency and magnitude to increased potential for wind erosion is highly uncertain and remains a difficult challenge to aeolian researchers.

Soil Crusting. Another significant control for aeolian processes in rangelands is physical and biological soil crusting, which significantly increases transport thresholds (Gillette et al. 1982; Belnap and Gillette 1998). Few papers have focused specifically on projected climate change implications for soil crusts in the context of soil erosion (Ferrenberg et al. 2015), and this remains a significant knowledge gap in our understanding of how climate change will impact aeolian processes in US rangelands. However, some relationships are relatively well understood. Both types of crust reduce wind erosion and have variable effects on water erosion. Climate change is likely to affect the proportion of the soil surface covered by different types of crusts at any time through one or more of the following processes: reduced rainfall frequency reduces opportunities for physical crust reformation following physical disturbance, but increased intensity generates stronger crusts that may be more persistent. Warming and changes to rainfall timing and intensity may negatively affect biological crusts (Belnap and Eldridge 2003) at a level similar to physical disturbance (Ferrenberg et al. 2015). While crusts can withstand limited soil mobility (Langston and McKenna Neuman 2005; Kidron and Zohar 2014), even small increases in aeolian activity because of reductions in extent, duration, or strength of soil crusts could potentially lead

Figure 5

Conceptual model of hydrologic vulnerability and risk. Hydrologic vulnerability (measured as runoff and erosion response, y-axis) is a function of site susceptibility (x-axis) and rainfall intensity (indicated by colors). Vulnerability dictates resources at risk. Concentrated flow processes dominate the postfire environment where site susceptibility is high (e.g., high bare ground, water-repellent soils) or rainfall is of moderate to high intensity. Rainsplash processes prevail where susceptibility is low (well-aggregated soils, land surface well protected by litter cover). Rainsplash, sheetflow, and concentrated flow processes all contribute to the runoff/erosion response where site susceptibility is indicative of disturbed conditions (reduced ground cover and aggregate stability, poor soil structure). The overall hydrologic response is amplified with increasing slope steepness. Sources: modified from Pierson et al. (2011) and Williams et al. (2014a).



to further instability and increased susceptibility to wind erosion.

Rainfall Erosivity. Climate change models consistently project an increase in the frequency and intensity of high intensity rainfall events for the western United States (figure 4) (Easterling et al. 2017), and thus erosivity (Nearing 2001; Zhang et al. 2012; Biasutti and Seager 2015; Nearing et al. 2017). Rainfall erosivity is defined as the capability of rainfall to cause soil loss by water and is a function of the energy of rainfall and rainfall intensity (Nearing et al. 2017). However, Nearing et al. (2017) point out that kinetic energy of rainfall is not a direct driver of erosion in cases where runoff (transport) is limited, rather water erosion rates are generally highest where overland flow occurs and that rainfall energy is effectively a surrogate for discharge or runoff generation. Runoff rates are generally greatest where surface conditions are bare (high susceptibility) and, under these conditions, are exacerbated with increasing rainfall intensity—high vulnerability (figure 5) (Pierson et al. 2011; Williams et al. 2014b; Pierson and Williams 2016).

Magnitude of erosion under these conditions is dependent on the connectivity of runoff; the amount of sediment available for transport; and the amount, energy, and transport capacity of overland flow (Pierson et al.

2011; Al-Hamdan et al. 2012, 2013, 2015; Williams et al. 2016a). Well-connected, bare conditions and high sediment availability are most common following short-term disturbances such as wildfire, but also occur with declines in vegetation and cover over time because of poor land use practices, drought, and woody plant encroachment (Pierson et al. 2007, 2010; Turnbull et al. 2008, 2012; Al-Hamdan et al. 2012; Williams et al. 2014a, 2016b, 2016a). Given these relationships, climate change impacts on erosivity are likely compounding in their effect on erosion through concomitant changes in plant community dynamics and disturbance regimes such as fire, drought, and desertification. As such, increases in rainfall erosivity may have the greatest impact through exacerbating the effects of fire, desertification, and poor land use practices on runoff generation and soil loss by water (Zhang et al. 2012). It is also worth noting that during extreme weather events, the effect of high winds on rain (wind-driven rain) has the potential to significantly increase rainfall erosivity for a given precipitation intensity (Marzen et al. 2017). Given the projected increase in extreme weather events, this impact should not be ignored in research and modeling efforts.

Impacts on Disturbance. The frequency of large fires, annual area burned, and

drought-induced tree die-off have increased across the western US rangeland-dry forest continuum in recent decades and are expected to increase further with projected climate change (Breshears et al. 2005, 2009; Westerling et al. 2006; Keane et al. 2008; Morgan et al. 2008; Flannigan et al. 2009; Littell et al. 2009; Spracklen et al. 2009; Abatzoglou and Kolden 2011; Miller et al. 2011; Balch et al. 2013; Williams et al. 2014b). Much of the interior western United States now exists in a state in which rangeland and woodland wildfires stimulated by annual grasses and dense woody fuels have a greater likelihood of progressing upslope into dry forests, where wildfire activity is also increasing (Keane et al. 2008; Nelson and Pierce 2010; Balch et al. 2013).

Invasion by and expansion of annual cheatgrass is the primary cause of increased fire frequency and annual area burned on sagebrush rangelands throughout much of the western United States (Keane et al. 2008; Miller et al. 2011; Balch et al. 2013). Cheatgrass escalates wildfire activity on rangelands by increasing the horizontal continuity of fuels and the likelihood of ignition (Brooks et al. 2004; Link et al. 2006). Burning of cheatgrass infested sites favors cheatgrass dominance and perpetuates a recurring grass-fire cycle (Knapp 1996; Brooks et al. 2004; Davies et al. 2012; Balch et al. 2013). Abatzoglou and Kolden (2011) suggest both cheatgrass invasibility and the length of the fire season in the Great Basin will be enhanced by a warmer climate and an increase in wet winters. Across the interior west, cheatgrass is moving upslope (Keeley and McGinnis 2007; McGlone et al. 2009; Griffith and Loik 2010; Bromberg et al. 2011), potentially setting up occurrence of the cheatgrass-fire cycle at higher elevations and in woodlands and dry forests. At mid-elevations, pinyon (*Pinus* spp.) and juniper (*Juniperus* spp.) expansion into sagebrush (*Artemisia* spp.) communities and infill on existing woodlands have increased woody fuel loading and increased risk of severe wildfires (Keane et al. 2008; Romme et al. 2009).

In recent decades, warming winter and spring air temperatures at mid-elevations of the western United States have yielded decreased snowpacks (Mote et al. 2005; Regonda et al. 2005; Knowles et al. 2006; Trenberth et al. 2007; Bonfils et al. 2008; Nayak et al. 2010), earlier spring snowmelt and streamflow (McCabe and Clark 2005; Regonda et al. 2005; Stewart et al. 2005; Pederson et al. 2011), and drier fuel condi-

tions (Westerling et al. 2006). These trends have extended fire seasons and increased fire frequency and area burned in western forests (Pierce et al. 2004; Westerling et al. 2006; Morgan et al. 2008; Pierce and Meyer 2008; Littell et al. 2009). In the southwestern United States, recent landscape- to regional-scale die-offs of pinyon and juniper have been attributed to periods of drought and limited soil water availability, plant water stress, bark beetle infestations, and reduced tree regeneration (Allen 2007; Allen and Breshears 1998; Breshears et al. 2005, 2009; Clifford et al. 2013; Gaylord et al. 2013; Redmond et al. 2013, 2015). Projections of climate and plant community transitions are highly variable, but most forecast warming and drought, increased dry-season cyclonic storms, longer fire seasons, and greater wildfire activity across the rangeland-dry forest continuum of the western United States (Flannigan et al. 2000, 2009; Whitlock et al. 2003; Gedalof et al. 2005; Running 2006; Bradley et al. 2009; Spracklen et al. 2009; Littell et al. 2010; Abatzoglou and Kolden 2011; Balch et al. 2013; Williams et al. 2014b). These broad-scale disturbances potentially increase susceptibility of rangelands to erosion across landscape to regional scales (Williams et al. 2014b).

Potential Trends and Implications of Soil Erosion

Increased aeolian sediment transport and dust emission have several implications for rangeland plant communities and ecogeomorphology and pose a serious hazard to human health. Soil loss and redistribution affects overall soil health, biogeochemical cycles, and land potential. Because many rangelands exist near ecological thresholds, even relatively small perturbations may degrade sites beyond their ability to recover (Archer and Predick 2008). Further, increased dust emissions could have significant implications for air quality and human health, while episodic hazards such as dust storms also pose immediate threats to human life.

Limited studies have attempted to quantify effects of climate change on wind erosion in US rangelands (Munson et al. 2011; Pu and Ginoux 2017). In general, lower projected wind speeds over much of the western United States (figure 2) will likely be offset by changes to vegetation controls, e.g., increases in intercanopy gaps. Further, more frequent fires and shorter fire return inter-

vals could lead to increased erosion from disturbed sites. Longer-term stresses and response to climatic change such as increased moisture deficits, decreased productivity, and grass-shrub transitions also have the potential to significantly increase site susceptibility to aeolian processes. These effects could potentially be exacerbated by future reductions in soil crusting. Indeed, dust emission rates have increased significantly over the past two decades (Hand et al. 2016), despite wind stalling over much of the area. Given the potential impacts of climate change on US rangelands, overall dust emission could continue to increase over both short- and long-term time scales despite the predicted decrease in overall wind speed over much of the western United States. Plot-scale research has explored the interactions between vegetation change and aeolian processes in US rangelands, but the impact of ecosystem changes for current and future regional dust emissions has not been established over large areas. It is incumbent on the rangeland research and management communities to monitor responses to these stresses across relevant thresholds of aeolian activity, as well as how these changes promote dust emission frequency and magnitude.

Projected climate changes and vegetation community response suggest increased landscape vulnerability to soil erosion by water over much of US rangelands, with the most pronounced impacts likely related to drought and increased disturbance. Knowledge regarding drought impacts on erosion in US rangelands comes primarily from studies in the Southwest, which suggest that community transitions that result in a reduction of herbaceous cover and favor connected, bare intercanopy spaces can lead to increased runoff and erosion (Wilcox et al. 1996; Allen and Breshears 1998). These results suggest that, overall, prolonged vegetation reductions associated with drought may cause increases in erosion through increased surface susceptibility. In general, amplified runoff and associated erosion are likely during high water input, erosive events at sites where cover is decreased by any disturbance (Pierson et al. 2011; Williams et al. 2014b). Indeed, there is ample evidence from the Southwest that documents increased soil erosion following coarsening of plant community structure via grassland-shrubland conversion—e.g., desertification (Schlesinger et al. 1990; Turnbull et al. 2008, 2012)—associated with

a combination of factors including land use and climate (Buffington and Herbel 1965; Grover and Musick 1990; Bahre and Shelton 1993; Archer et al. 1995; Van Auken 2000, 2009). Coarsening of plant community structure increases fine-scale erosion by rainsplash and sheetflow (Abrahams et al. 1995; Parsons et al. 1996a). In turn, runoff generated in bare interspaces promotes concentrated flow and amplifies downslope sediment transport (Luk et al. 1993; Parsons et al. 1996b; Wainwright et al. 2000; Turnbull et al. 2010). Water and soil losses inhibit herbaceous productivity and further propagate bare ground connectivity (Bhark and Small 2003). In general, both wind and water erosion increase with increasing bare ground over broad scales, potentially degrading sites beyond conservation and restoration thresholds (Turnbull et al. 2012). Similar responses have been reported for coarsening of plant community structure following shrubland-to-woodland conversions driven by land use, climate, and reduced fire activity in the northwestern United States (Pierson et al. 2007, 2010, 2013; Williams et al. 2014a, 2016b, 2016c, 2016a). In contrast, recent studies of streamflow following drought-induced pinyon, juniper, and lodgepole pine (*Pinus contorta* Douglas) die-off in the Southwest have reported either decreases to little change in streamflow over short timescales (Guardiola-Claramonte et al. 2011; Biederman et al. 2015). In both studies, the authors suggest that decreases in streamflow were associated with increased water use by enhanced herbaceous cover following tree die-off. Thus, the limited research to date on drought-specific impacts on water erosion from US rangelands hinders accurate prediction of drought impacts on erosion with climate change, and this should be a research focus moving forward.

Disturbance by wildfires increases susceptibility and vulnerability to erosion at plot to watershed scales and increases hazards to values-at-risk such as natural resources, property, infrastructure, and human life (figure 5) (Calkin et al. 2007; Robichaud et al. 2010; Pierson et al. 2011; Williams et al. 2014b; Pierson and Williams 2016). Much of what is currently known about fire impacts on erosion comes from plot- to hillslope-scale artificial rainfall experiments and anecdotal reports of debris flows and mudslides (Pierson et al. 2002, 2011; Williams et al. 2014b; Pierson and Williams 2016). Increasing wildfire activity associated with climate change

poses significant environmental, social, and economic consequences associated with erosion events (Pierson et al. 2011; Wilcox et al. 2012; Williams et al. 2014b). More frequent and larger fires increase the likelihood and potential magnitude of both onsite and off-site impacts to values-at-risk (figure 5). More frequent surface exposure due to repeated burning will likely increase both water and wind erosion during commonly occurring storms in addition to increasing the probability susceptible soil surfaces will be exposed when less frequent, high intensity events—which climate projections indicate are likely to increase (Kossin et al. 2017)—occur. Loss of biologically important surface soils during more frequent high intensity events may be particularly critical for rangelands where soil formation takes decades (Allen et al. 2011; Sankey et al. 2012). This may be especially problematic where large fires are followed by drought years with minimal plant recruitment that prolongs susceptibility.

Waterborne sediment pulses transported into channels during postfire erosion events and subsequently flushed into streams (Cannon et al. 2001; Meyer and Pierce 2003; Pierce et al. 2004) may negatively impact water resources, fisheries, and channel geomorphology (Minshall et al. 2001; Pierce et al. 2011). Further, several studies have linked large debris flow events in dry forests of the Interior West to warm climatic conditions (Medieval Warm Period, 1,050 to 750 years ago) and periods of extensive fires (Meyer and Pierce 2003; Pierce et al. 2004; Pierce and Meyer 2008). Fire-induced debris flows can transport tremendous volumes of sediment and debris into main stem rivers (Cannon et al. 2001; Meyer et al. 2001; Pierce et al. 2011), and the literature suggests that these events may become more common under continued increases in wildfire activity and high intensity, erosive rainfall events on extensively burned landscapes (Williams et al. 2014b).

Current knowledge is strong regarding fire impacts on surficial processes across plot to small watershed scales, but research is still needed to relate varying surface susceptibility to runoff and erosion responses for burned versus unburned conditions and to better represent hydrologic and erosion recovery in quantitative hydrology and erosion models under changing precipitation regimes (Pierson et al. 2011; Moody et al. 2013; Williams et al. 2014b; Pierson

and Williams 2016). Likewise, current ability to accurately model and predict postfire hydrologic and associated erosion responses over large spatial scales and complex topography remains limited, particularly under potentially changing precipitation regimes and erosivity and for flood, debris flow, and mud-slide responses. More work is also needed to assess the impacts of fire on aeolian transport rates and dust emission events (Dukes et al. 2018). Advancements in these key areas are important in the development of postfire decision support tools for assessing and predicting erosion responses under a changing climate.

Discussion

Managing the impacts of climate change on wind and water erosion will be critical for the long-term sustainability of US rangelands (Webb et al. 2017). Our review of potential climate change impacts on erosion drivers, and the likely responses, brings us to some important considerations for the management of current and future erosion as the basis for maintaining and improving soil health, water and air quality, and the ecosystem services that depend on them. Foremost are the following questions:

1. How can wind and water erosion and their responses to land use and management be effectively monitored across diverse rangeland ecosystems?
2. How can information about soil erosion be incorporated into management decisions?
3. What management actions are necessary to reduce the negative impacts of climate change while leveraging positive impacts?
4. What proportion of often limited management resources should be devoted to anticipating future climate change impacts versus addressing current impacts of invasive species, grazing-induced changes in the fire regime, and soil degradation?

Quantifying wind and water erosion rates and their responses to land use, land management, and land cover changes will be essential to develop management responses that are effective into the future. Currently, our understanding of regional wind erosion rates is limited and supported largely by coarse spatial resolution (e.g., $>1^\circ$) model estimates of dust emission (Shao et al. 2011). Little is known about the current magnitude of wind erosion rates in US rangelands (Breshears et al. 2003; Nauman et al. 2018), with most available measure-

ments of sediment transport rather than erosion, and with limited scope (Munson et al. 2011; Wagenbrenner et al. 2012; Webb et al. 2014). More is known about runoff and water erosion in US rangelands because of its longer research focus (Nearing et al. 2017). As measuring wind and water erosion rates over the large area of US rangelands is impractical, modeling approaches that can leverage data collected by existing monitoring programs—or remote sensing (Chappell et al. 2018)—provide the best opportunities for evaluating erosion rates and the process responses to land use and climate change. The Aeolian EROsion (AERO) model (Edwards et al. 2018) and Rangeland Hydrology and Erosion Model (RHEM) (Nearing et al. 2011; Al-Hamdan et al. 2015) can be applied to standardized rangeland monitoring data—like those collected as part of the Bureau of Land Management’s Assessment, Inventory and Monitoring (AIM) program (Toevs et al. 2011) and the USDA Natural Resources Conservation Service’s National Resources Inventory (NRI) (Goebel 1998)—and provide the opportunity to assess rangeland wind and water erosion across plot to regional scales. As of December of 2017, AIM and NRI data have been collected at over 50,000 locations across the US rangelands to which AERO and RHEM can be applied, with support in model calibration and testing to establish uncertainties from networked observatory sites like the USDA’s Long-Term Agroecosystem Research Network (Robertson et al. 2008) and associated National Wind Erosion Research Network (Webb et al. 2016).

Establishing model estimates of water erosion and wind-driven sediment transport from existing monitoring data will provide new opportunities to link erosion assessments to land classification systems and land resource hierarchies that are used to inform management (Salley et al. 2016). For example, incorporating model estimates of wind and water erosion from AIM and NRI data into ecological site descriptions (ESDs) (Caudle et al. 2013) and relating erosion rates to core indicators of rangeland condition (Herrick et al. 2017) and rangeland health (Pyke et al. 2005) could provide the necessary information for resource managers to identify tolerable erosion rates; establish benchmarks (e.g., in ground cover) for erosion control, which relate to water and air quality (e.g., 1977 Clean Air Act and National Ambient

Air Quality Standards); and to consider erosion management and trade-offs alongside other resource concerns. Recent applications of ESDs and state-and-transition models to evaluate rangeland dust emission (Galloza et al. 2018) and water erosion (Williams et al. 2016b, 2016c) have shown the utility of integrating monitoring data with erosion models and conceptual models of ecosystem states and their dynamics (figure 6). Such systems approaches will be useful for understanding and managing the impacts of climate change on rangeland erosion and feedbacks of the processes to ecosystem health and production.

We contend that management actions and resources needed to address future climate change on soil erosion in rangelands are largely those necessary to address the current impacts of invasive species, fire, and soil degradation. Increasing the health and overall resilience of rangeland ecosystems will in many, if not most, cases reduce negative climate change impacts and improve the efficacy of climate adaptation while allowing positive impacts to be exploited (Webb et al. 2013). At a minimum, an increased focus on addressing or adapting to current threats—e.g., shrub invasion of Chihuahuan Desert ecosystems (Archer et al. 2017), changing grass-fire cycles and invasion of Great Basin sagebrush shrublands by cheatgrass (Balch et al. 2013), and slow ecological recovery of ecosystems affected by infrastructure development for oil and gas extraction (Nauman et al. 2017)—that are likely to be *exacerbated* by climate change and impact rangeland communities is justified (Dallimer and Stringer 2018). Planned rather than reactive management of rangeland soil erosion and the potential process responses to climate change is likely to have the greatest positive impact on the quality of rangeland soil, vegetation, air, and water resources over the long term (Briske et al. 2015). Participatory research to develop innovative range management systems that are inherently more flexible and suited to changing climatic and market conditions are also likely to build resilience to an increasingly unpredictable climate (Spiegel et al. 2018).

Finally, among the many climatic controls on erosion, our confidence in their midcentury projections vary dramatically. There is relatively high confidence in projections of rising temperatures, as well as more frequent and severe soil moisture deficits, wildfires, and heavy precipitation

events. However, confidence is relatively low in projections of average precipitation (annual and seasonal), intense storms, and surface wind speeds. Changes in land use and land cover may be equally if not more important as climate change in determining future erosion rates (Ferrenberg et al. 2015) but may well be influenced by climatic changes. There is great uncertainty about how plant communities will respond to climate change and how the socioeconomic drivers of land use change will impact rangeland wind and water erosion. In the face of such uncertainty, explicitly considering the trade-offs for erosion impacts against other management objectives and implementing flexible, “no-regrets” management practices that achieve multiple objectives could have both immediate and long-term benefits for US rangelands.

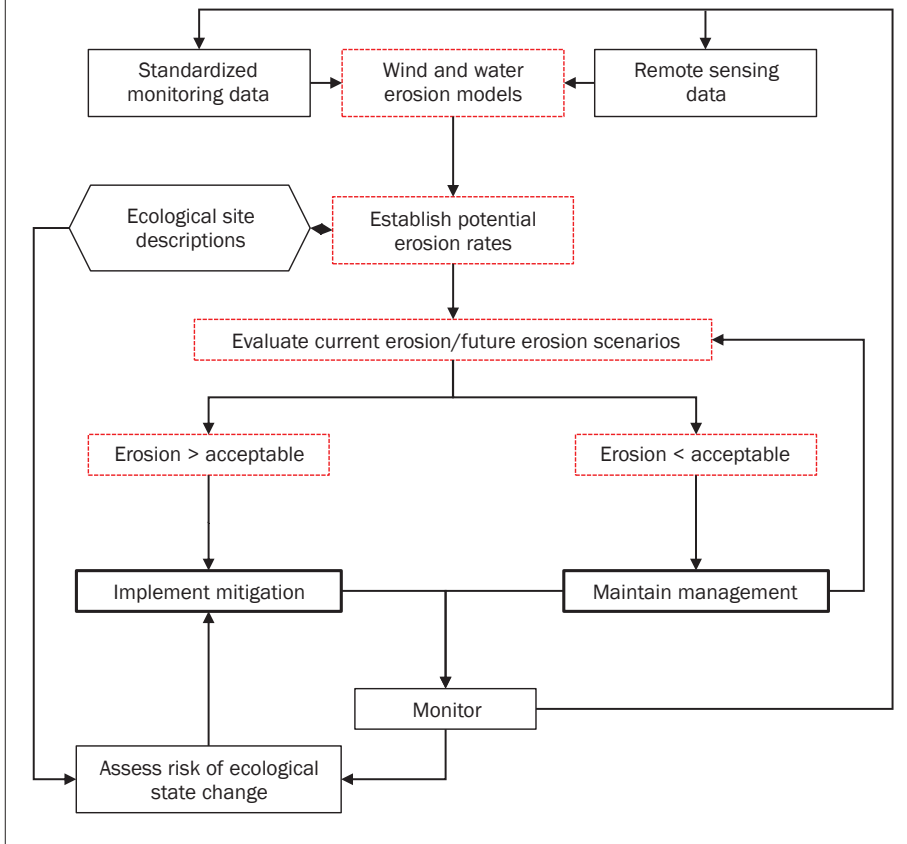
Summary and Conclusions

Conservation of US rangeland resources into the future presents a substantial challenge to the management community. Soil erosion by wind and water is already a serious threat to US rangelands and may be accelerated by future climate change. Managing to limit soil erosion impacts today will be critical for adaptation, given that many of the challenges we currently face are expected to be exacerbated if current trajectories of emissions and climate change continue. Moreover, given our current understanding, more widespread erosion will only increase exposure and sensitivity of rangelands to climate impacts—reducing resilience to climatic stressors and pushing some rangelands past restoration and conservation thresholds—which reduces the adaptive capacity of land users. Our understanding of ecogeomorphic links between erosion processes and rangeland condition and trajectory is improving, but more research is needed to inform management efforts to build rangeland resilience. The following summarize our current understanding of the potential impacts of climate change on erosion in US rangelands and highlight important knowledge gaps and research needs:

- Projected trends toward reduced vegetation cover and shifts in vegetation communities from herbaceous to shrub-dominated landscapes resulting from warming and drying have the potential to significantly increase landscape susceptibility to water and wind

Figure 6

Generalized schema that includes accounting for current erosion in evaluating and managing rangeland soil planning for future climate changes.



soil erosion over much of the western United States.

- Potential increases in rainfall erosivity may have the greatest impact by exacerbating effects of fire, desertification, and poor land use practices on runoff and erosion.
- Increased disturbance from fire and exotic invasive grasses has the potential to significantly increase frequency and magnitude of erosive events over both short time scales commensurate with increased susceptibility during recovery periods and longer time scales associated with increased vulnerability to erosion because of more frequent, larger fires over an expanded range.
- Projected climate change could slow recovery times to fire and other disturbances, e.g., infrastructure development for oil and gas extraction, that are increasing in density across western rangelands and may impact soil erosion and dust emission.
- More work is needed to expand our ability to model postfire hydrologic and

erosion response over large areas and complex topography.

- More research is needed to understand the impact of ecosystem changes, e.g., reductions in vegetation density, community shifts, and soil crusting, for current and future wind erosion and regional dust emission from US rangelands.
- More research is needed to improve understanding of drought-specific impacts on water erosion in US rangelands, which is currently limited.
- Existing, extensive plot-scale rangeland monitoring and remote sensing data should be leveraged in modeling frameworks to establish baseline assessments of current vulnerability of US rangelands to soil erosion and, where possible, used to assess potential changes to future vulnerability associated with rangeland responses to climate change.
- Soil erosion should be explicitly included in management benchmarks and decision support tools to better mitigate negative consequences of erosion and its impacts on water resources and air quality.

- Flexible, no-regrets management options should be implemented to combat current soil erosion impacts on US rangelands, which will improve rangeland resilience and the efficacy of future climate change adaptation.
- Finally, fundamental knowledge of erosion processes is critical to developing accurate models and appropriate management practices. As such, process-based research to further our understanding of the underlying processes and refine models used to guide management is a continual need.

Acknowledgements

We would like to thank Paige Ramsey (GIS specialist at the USDA Southwest Climate Hub in Las Cruces, New Mexico) for her patient assistance with producing figures for this article.

Disclaimer

This work is a product of the USDA Agricultural Research Service. Any opinions, findings, conclusions, or recommendations expressed in this publication are those of the author(s) and do not necessarily reflect the view of the USDA, which is an equal opportunity provider and employer. Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the US Government.

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