3. CLIMATE ADAPTATION

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Introduction

Management actions that enable adaptation to climate change and promote resilience to disturbance are becoming increasingly important in the sagebrush biome. In recent decades temperatures have increased, growing seasons have lengthened, and in many areas the timing and amount of precipitation has changed (Chambers et al. 2017 [hereafter, Part 1], section 4; Kunkel et al. 2013a,b,c). Global climate change models are used to project future changes in temperature and precipitation based on relative concentration pathways of likely emissions of carbon dioxide ($CO_{2)}$ and other trace gases and information on the Earth's surfaces and oceans. These models project continued temperature increases and additional changes in precipitation throughout the remainder of the century, although the magnitude and rate of change differ based on the relative concentration pathway used (Part 1, section 4; Kunkel et al. 2013a,b,c).

Continued changes in climate are likely to influence the distributions of native species (Bradley 2010; Homer et al. 2015; Schlaepfer et al. 2012c; Still and Richardson 2015), invasive annual grasses (Bradley et al. 2016), fire regimes (Abatzoglou and Kolden 2013; Littell et al. 2009; Westerling et al. 2014), and insects and disease (Bentz et al. 2016). Snowpacks are declining in many areas (Mote and Sharp 2016), droughts are becoming more severe (Cook et al. 2015; Prein et al. 2016), and the length of the fire season and duration of extreme fire weather is increasing (Abatzoglou and Kolden 2013; Littell et al. 2009; Westerling et al. 2014; but see also McKenzie and Littell 2017). Reducing ecosystem vulnerability, or the degree to which a system is susceptible to the adverse effects of climate change, including climate variability and extremes (IPCC 2014), will require scientific guidance and agency direction to enable climate adaptation planning and implementation across scales.

Climate adaptation, the process of adjusting to actual or expected changes in climate, is an important consideration when developing management strategies in the face of climate change. The focus of climate adaptation is to moderate or avoid harm or to exploit beneficial opportunities (IPCC 2014). Adaptation can be **incremental**, where the objective is to maintain the integrity of a system or process at a given scale. Climate scientists anticipate that climate will continue to change throughout the 21st century due to continued accumulation of greenhouse gases in the atmosphere. As the climate warms, ecosystems may not persist in their current locations. Thus, adaptation can also be **transformational**, where actions focus on changing the fundamental attributes of a system in response to climate and its effects (IPCC 2014). Mitigation of climate change is another approach to managing climate change that is based on reducing the sources or enhancing the storage of greenhouse gases (IPCC 2014). This section focuses on incremental and transformational adaptation actions that can enhance the resilience of sagebrush systems. It also reviews the available information on the effects of management actions on carbon storage.

Top: Road to Nixon, Nevada, sunrise (photo by Nolan Preece, used with permission). Middle right: Dr. Matt Germino illustrating a weather station on the Soda Fire in SE Idaho (photo: U.S. Geological Survey). Middle left: A common garden study for assessing the importance of local adaptation in sagebrush (photo: USDA Forest Service). Bottom: Planting sagebrush seedlings after a wildfire (photo: USDA Forest Service). Bottom inset. Sagebrush transplant (photo: Stacy Moore, Institute for Applied Ecology).

Climate Adaptation and Resilience Management

Concepts

Managing natural resources within the context of climate adaptation is consistent with the approach described in Part 1 of the Science Framework, but requires the necessary flexibility to modify management actions as environmental conditions change. Widely used concepts for addressing adaptation in use by the Fish and Wildlife Service (FWS) (USDOI FWS 2010), the Forest Service (USDA FS 2011), and their partners focus on climate resistance, resilience, response, and realignment strategies (Halofsky et al. 2018a,b). Resistance strategies aim to increase the capacity of ecosystems to retain their fundamental structure, processes, and functioning despite climate-related stressors such as drought, wildfire, insects, and disease. These types of strategies may offer only short-term solutions, but often describe the intensive and localized management of rare and isolated species (Heller and Zavaleta 2009). Strategies to increase ecosystem resilience aim to minimize the severity of climate change impacts by reducing vulnerability and increasing the capacity of ecosystem elements to adapt to climate change and its effects. Response strategies seek to facilitate large-scale ecological transitions in response to changing environmental conditions and may include realignment or the use restoration practices to ensure persistence of ecosystem processes and functions in a changing climate.

These concepts of climate resistance, resilience, and response apply to many management and land ownership contexts and can be used to help determine appropriate climate adaptation strategies. Using these concepts to manage for changes in climate involves examining whether current assumptions about the effects of weather and climate on environmental responses and underlying assumptions about the expected result of management actions are still viable in a changing environment. Examples are ecological site descriptions and state-and-transition models in which the reference state often serves as the management target (fig. 3.1) (Bestelmeyer et al. 2009; Briske et al. 2005; Caudle et al. 2013). While managers can use historical data to help understand ecosystem response to environmental changes (e.g., Swetnam et al. 1999), it is important to recognize that the relationship between climate and ecosystem response will shift over time with continued warming. Consequently, managing for historical conditions may not maintain ecological sustainability (goods and services, values, biological diversity) into the future and management actions should be planned accordingly (Hobbs et al. 2009; Millar et al. 2007).

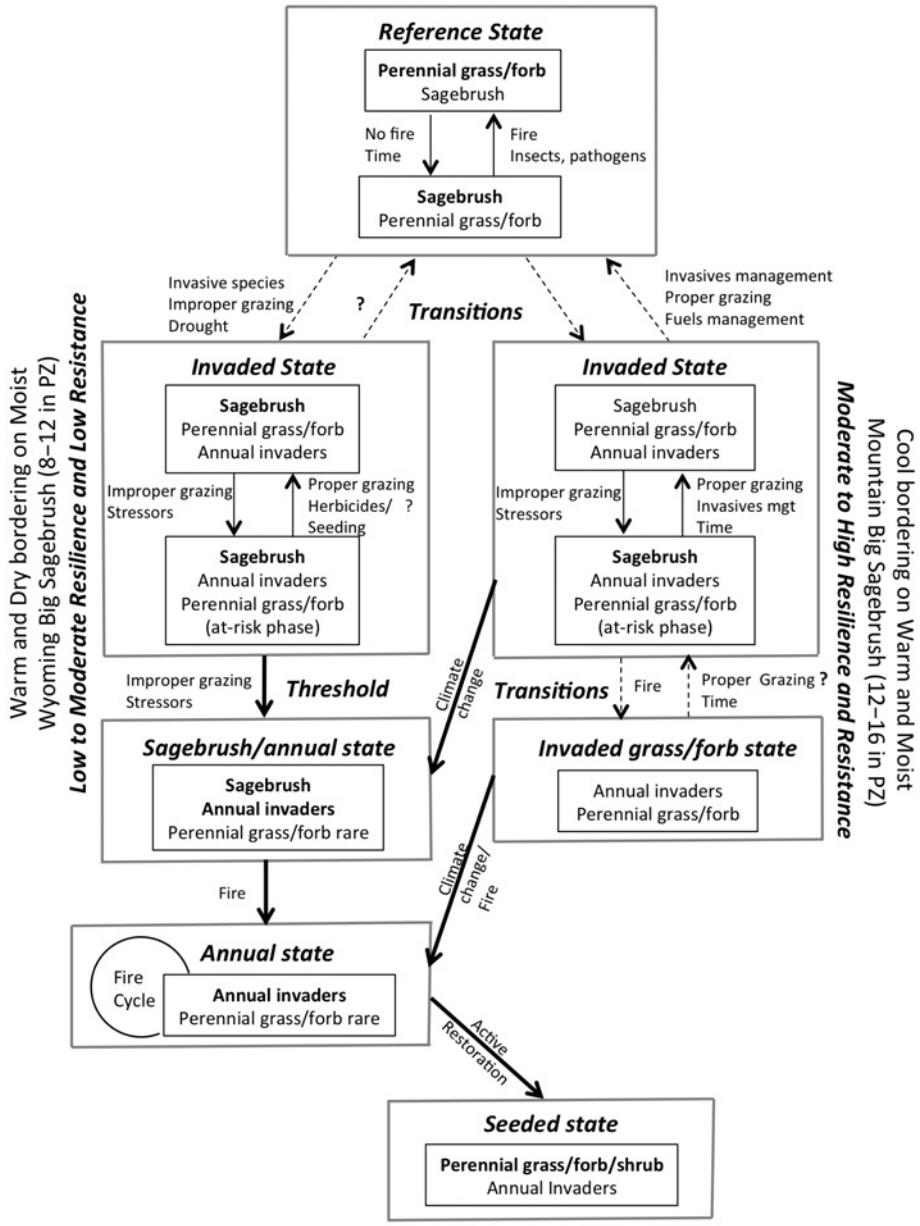

Figure 3.1—Generalized conceptual model showing the states, transitions, and thresholds for relatively warm and dry Wyoming big sagebrush ecosystems with low to moderate resilience and low resistance to cheatgrass and cool and moist mountain big sagebrush ecosystems with moderate resilience and resistance in the Cold Deserts (Chambers et al. 2017, Appendix 6). Reference state: Vegetation dynamics are similar for both types. Perennial grass/forb increases due to disturbances that decrease sagebrush, and sagebrush increases with time after disturbance. Invaded state: An invasive seed source, improper grazing, stressors such as drought, or a combination thereof, trigger a transition to an invaded state. Perennial grass/forb decreases, and both sagebrush and invaders increase with improper grazing and stressors, resulting in an at-risk phase in both types. Proper grazing, invasive species management, and fuel treatments may restore perennial grass and decrease invaders in relatively cool and moist Wyoming big sage and in mountain big sage types with adequate grass/forb, but return to the reference state is likely only for mountain big sage types. Sagebrush/annual state: In the Wyoming big sagebrush type, improper grazing and stressors trigger a threshold to sagebrush/annual dominance. Annual state: Fire, disturbances, or management treatments that remove sagebrush result in dominance of annuals. Perennial grass is rare, and repeated fire causes further degradation. Seeded state: Active restoration results in dominance of perennial grass/forb/shrub. Treatment effectiveness and return to the annual state are related to site conditions, posttreatment weather, and seeding mixture. Invaded grass/forb state: In the mountain big sagebrush type, fire results in a transition to annual invaders and perennial grass/forb. Proper grazing and time may result in return to the invaded state given adequate perennial grass/forb. Increases in climate suitability for cheatgrass and other annual invaders may shift vegetation dynamics of cooler and moister mountain big sagebrush ecosystems toward those of warmer and drier Wyoming big sagebrush ecosystems. Although not shown here, woodland expansion and infill in mountain big sagebrush sites with conifer potential can result in transition to woodland-dominated or eroded states, leading to crossing of biotic and abiotic thresholds (adapted from Chambers et al. 2014b).

Climate Adaptation Strategies

Due to uncertainty about exactly what the future will look like, planning for multiple possibilities and using adaptive management principles is essential. Adaptive management uses the best available information for helping ecosystems and the plant and animal species they support to adapt to inevitable changes in climate (Millar et al. 2007). Climate adaptation strategies for the sagebrush biome are in table 3.1. The specific approaches for sagebrush ecosystems build on the sage-grouse habitat resilience and resistance matrix (table 1.3) and the sagebrush ecosystem management strategies (table 1.4).

Climate adaptation strategies incorporate multiple scales and focus on preventing the loss of ecosystem services by maintaining and enhancing ecosystem processes, functional attributes, and feedbacks (table 3.1). For example, the extent and connectivity of intact sagebrush ecosystems provide a buffer that facilitates species adaptation and movement in response to climate change as well as to the increasing effects of human development and land use (e.g., Knick et al. 2011, 2013; Millar et al. 2007). Maintaining intact and connected sagebrush ecosystems is based on developing public land use plans (LUPs) and policies that reduce the impact of existing ecological, land use, and development stressors on these ecosystems at broad (sagebrush biome and multiple Management Zones) to mid- (Greater sage-grouse [*Centrocercus urophasianus*; hereafter, GRSG] Management Zone and ecoregion) scales. It also involves strategic placement of conservation easements to prevent conversion to tillage agriculture and anthropogenic developments and to maintain existing connectivity at mid- to local (district, field office, or project level) scales.

Many climate adaptation strategies work together to accrue multiple ecosystem benefits. Maintaining or enhancing key plant structural and functional groups is central to most climate adaptation strategies. Certain plant structural and functional groups are critical for stabilizing hydrologic and geomorphic processes, promoting desired successional processes, and lowering risk of conversion to invasive annual grasses following disturbances that remove native vegetation (Pyke 2011). Postfire rehabilitation and restoration activities can increase ecosystem capacity to absorb change by using functionally diverse species mixtures and including plant materials from across a greater geographic range that considers current climate and near-future climate (next 20 to 30 years) (table 3.1) (Butler et al. 2012; Finch et al. 2016). Favoring existing genotypes that are better adapted to future conditions because of broad tolerances to disturbances, drought adaptations, pest resistance, or other characteristics can also increase adaptive capacity (table 3.1) (Butler et al. 2012; Finch et al. 2016). Where shown to be successful, assisted migration, or the purposeful movement of individuals or propagules of a species to facilitate or mimic natural range expansion or long-distance gene flow within the current range, may facilitate community adjustments (Buchorava 2017). Implementing these strategies requires developing the necessary research and management capacity to forecast changes in ecological conditions and species distributions and to better understand ecosystem and species response to changes in climate at mid- to local scales.

Table 3.1–Climate change adaptation strategies for the sagebrush biome. General strategies are based on Millar et al. (2007, 2012) and Butler et al. (2012). Specific approaches for sagebrush ecosystems build on the sage-grouse habitat, resilience and resistance matrix (table 1.3) and management strategies for persistent ecosystem threats and land use and development threats (table 1.4). Resistance = R1; Resilience = R2; Response = R3.

Sustain fundamental ecological conditions (R1, R2, R3)
- Maintain or restore soil quality and nutrient cycling by reevaluating the timing and intensity of land use practices such as livestock grazing
- Maintain or restore hydrologic and geomorphic processes following stress and disturbance

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Reduce the impact of existing ecological, land use, and development stressors (R1, R2, R3)
- Develop appropriate policies, land use plans, and project plans to protect sagebrush habitat and prevent fragmentation
- Secure conservation easements to prevent conversion to tillage agriculture, housing developments, and other land conversions, and maintain existing connectivity

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Promote landscape connectivity (R2, R3)
- Reduce juniper and piñon expansion to maintain connectivity among sage-grouse and sagebrush dependent species populations and facilitate seasonal movements
- Suppress fires that occur under more severe burning conditions in targeted areas where altered fuel beds facilitate large and severe fires, increase landscape fragmentation, and impede dispersal, establishment, and persistence of native plants and animals
- Manage landscapes to create or enhance permeability and increase the ability of sagebrush dependent species to move between individual Priority Areas for Conservation or Biologically Significant Units

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Maintain or create refugia (R1)
- Identify and maintain ecosystems that: (1) are on sites that may be better buffered against climate change and short-term disturbances, and (2) contain communities and species that are at risk across the greater landscape
- Prioritize and protect existing populations on unique sites
- Prioritize and protect sensitive or at-risk species or communities
- Establish artificial reserves for at-risk and displaced species

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Reduce the risk of wildfires that result in abrupt transitions to novel states (R1, R2)
- Reduce fuel loads and fuel continuity to (1) decrease fire size, alter burn patterns, decrease perennial grass mortality, and maintain landscape connectivity; (2) decrease competitive suppression of native perennial grasses and forbs by woody species, including sagebrush; and thus (3) lower the longer-term risk of dominance by invasive annual grasses and other invaders
- Use mechanical treatments (e.g., cutting, mastication) to reduce woody fuels in juniper and piñon expansion areas with moderate to high resilience that have little or no presence of invasive annual grasses and sufficient perennial grasses and forbs to promote recovery
- Use prescribed fire to create fuel mosaics and promote successional processes in sagebrush and juniper and piñon expansion areas with moderate to high resilience that have little or no presence of invasive annual grasses and sufficient perennial grasses and forbs to promote recovery
- Use herbicide applications and appropriately timed livestock grazing to reduce cheatgrass fuels in sagebrush ecosystems where they have potential to increase perennial grasses and forbs
- Suppress wildfires in moderate and especially low resilience and resistance sagebrush-dominated areas to prevent conversion to invasive annual grass states and thus maintain ecosystem connectivity, ecological processes, and ecosystem services
- Suppress wildfires adjacent to or within recently restored ecosystems to promote recovery and increase capacity to absorb future change
- Use fuel breaks in carefully targeted locations along existing roads where they can aid fire suppression efforts and have minimal effects on ecosystem processes (Maestas et al. 2016)

Reduce the risk of nonnative invasive plant species introduction, establishment, and spread (R1, R2, R3)
- Limit anthropogenic activities that facilitate invasion processes including surface disturbances, altered nutrient dynamics, and invasion corridors
- Use Early Detection and Rapid Response (USDOI 2016) for emerging invasive species of concern to prevent invasion and spread
- Manage livestock grazing to promote native perennial grasses and forbs that compete effectively with invasive plants
- Actively manage invasive plant infestations using integrated management approaches such as chemical treatment of invasives and seeding of native perennials from climatically appropriate seed sources

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(Continued)

Table 3.1—(Continued).

Maintain or enhance key structural and functional groups (R1, R2, R3) • Manage grazing by livestock and wild horse and burro populations to maintain soil and hydrologic functioning and capacity of native perennial herbaceous species, especially perennial grasses, to effectively compete with invasive plant species • Manage grazing by livestock and wild horse and burro populations to maintain riparian-wetland functioning, streambank and floodplain stability, and vegetation sufficient to dissipate flood energy, promote infiltration, minimize erosion, and compete with invasive plant species • Reduce conifer expansion to prevent high severity fires and maintain native perennial herbaceous species that can stabilize geomorphic and hydrologic processes and minimize invasions • Restore disturbed areas with functionally diverse mixtures of native perennial herbaceous species and shrubs with climatically appropriate seed sources and with capacity to persist and stabilize ecosystem processes under altered disturbance regimes and in a warming environment
Enhance genetic diversity (R2, R3) • Use seeds, germplasm, and other genetic material from across a greater geographic range based on current climate and near-future (next ~20–30 years) climate considerations • Favor existing genotypes that are better adapted to future conditions because of pest resistance, broad tolerances, or other characteristics • Increase diversity of nursery stock to provide those species or genotypes likely to succeed
Facilitate community adjustments through species transitions (R3) • Monitor both native and invasive species at range margins to provide advanced warning of range shifts • Investigate assisted migration options—the purposeful movement of individuals or propagules of a species to facilitate or mimic natural range expansion or long-distance gene flow within the current range—in areas with high rates of climate change
Plan for and respond to disturbance (R3) • Practice drought adaptation measures, such as altered grazing seasons or reduced grazing during droughts, and implement conservation actions to facilitate species persistence • Identify current and potential future areas where snowpack cover and duration are declining in order to manage to reduce other current stressors • Anticipate and respond to species declines such as may occur on the southern or warmer edges of their geographic range by including plant materials from neighboring climate types in seed and planting mixes • Leverage topographic features (landforms) that retain soil moisture longer for restoration activities (Bainbridge 2007) • Favor or restore native species that are expected to be better adapted to the future range of climatic and site conditions • Protect future-adapted restoration and reclamation seedings from inappropriate livestock grazing and wild horse and burro populations • Avoid seeding introduced forage species such as crested wheatgrass that outcompete native species (Davies et al. 2013; Lesica and Deluca 1996)

Management and research studies coupled with landscape monitoring can provide the basis for developing cost-effective and feasible management strategies for adapting to climate change. Carefully designed management and research studies implemented in the near future may increase our understanding of viable approaches for adaptation measures, such as appropriate grazing regimes for drought conditions, conservation actions to facilitate species persistence during climate warming, seeding and transplanting techniques during drought, and identification of species and ecotypes that can be used successfully in assisted migration. Monitoring to detect the rates and magnitudes of change occurring within the context of adaptive management can identify both populations and habitats that are declining (Carwardine et al. 2011; Field et al. 2004), as well as new or novel combinations of species that constitute a functioning ecosystem under climate change. Increased understanding of both the changes occurring and viable strategies for addressing those changes may reduce uncertainty and provide direction for adaptive management strategies (Hobbs et al. 2009).

A participatory scenario planning process may be one approach to help identify relevant adaptation strategies in the context of adaptive management (Cross et al. 2013; Star et al. 2016; USDA FS 2012; USDOI NPS 2013). Participants can use climate change projections and associated natural resource models to depict both the amount of change and the degree of uncertainty (Star et al. 2016). Decisionmakers, stakeholders, and experts can work together to identify the most relevant and uncertain drivers of system change, which often include sociopolitical and socioeconomic factors. They can then develop a shared understanding of future climate scenarios that is likely to lead to broader support for suggested adaptation strategies (Star et al. 2016). To date, scenario planning has been more commonly used in nongovernmental organizations and local government planning.

Prioritizing Management Actions and Determining Appropriate Management Strategies

Assessing ongoing and projected climate change using the best available data is integral to evaluating priority areas for management at mid-scales and determining appropriate management treatments at local scales. In the context of the Science Framework, the effects of changes in climate on species and ecosystems can be addressed similarly to other persistent ecosystem threats such as wildfire and invasive annual grasses (see Part 1, section 8; table 3.1, this volume). For GRSG and other at-risk species and resources, the process involves overlaying key data layers in a geospatial analysis to both visualize and quantify: (1) species locations and abundances, (2) the probability that an area has suitable habitat, (3) the likely response to disturbance or management treatments, and (4) the dominant threats including projected climate change.

Geospatial analyses with overlays of key data layers can: (1) help evaluate the level of risk to vegetation types and species to climate change, (2) target areas for adaptive management, and (3) determine the most appropriate types of management actions. Key data layers include projected changes in climate variables (Part 1, section 8). Land managers can use these layers to assess the rate and magnitude of change projected for the assessment area. Other important layers are projections for changes in individual plant species (e.g., Bradley et al. 2016; Homer et al. 2015; Still and Richardson 2015) and vegetation types (e.g., Rehfeldt et al. 2012; Schlaepfer et al. 2012c) under different climate change scenarios. In addition, climate change vulnerability analyses of key ecological and socioeconomic resources (water, fisheries, vegetation and disturbance, wildlife, recreation, infrastructure, cultural heritage, and ecosystem services) are available for the Intermountain Region (Halofsky et al. 2018b) and Northern Rocky Mountain Region (Halofsky et al. 2018a). Additional websites and resources for climate change are in Appendix 2.

Climate change projections can be factored into prioritizing areas for management within assessment areas (Part 1, section 8) by considering the following factors.

- Continued changes in climate (i.e., increases in temperature and shifts in precipitation timing and amount) and the associated effects are expected to be relatively small within the next decade or two. Areas can be prioritized for management that provide suitable habitat and support species populations at mid-scales, and management practices can be adapted to build resilience to changes in climate into sagebrush ecosystems at local scales (table 3.1). Monitoring can provide critical information on changes in species and ecosystems resulting from climate changes that allows managers to take advantage of opportunities to facilitate transitions to systems that will be better adapted in the long term.

- Changes in climate and the interactions of these changes with other threats are already documented and are expected to be large (e.g., rapid warming events, uncertainty of snowpack, extreme drought) in the next few decades (table 3.1). The impacts of changes in climate on plant community composition and vegetation types will be most evident following major disturbances, such as wildfires, that occur at an ecotone between different vegetation types or on warmer, drier sites. In this case, more proactive adaptation strategies may be necessary to facilitate community adjustments and species persistence. These may include favoring or restoring native species that have been shown to be better adapted to the future range of climatic and site conditions and to have acceptable effects on biotic interactions and ecosystem process (Bucharova 2017). The use of assisted migration to address changes in climate suitability will require additional research and management guidelines to evaluate the potential positive as well as negative effects of purposeful species movements (Bucharova 2017).

Key Topics in Climate Adaptation

Across much of the sagebrush biome, climate change is resulting in a warmer and drier environment (Kunkel et al. 2013a,b,c). In turn, the warmer, drier conditions are resulting in increasing magnitude and frequency of droughts (Cook et al. 2015; Prein et al. 2016), increasing dust in the atmosphere (Livneh et al. 2015; Painter et al. 2012; Steltzer et al. 2009), and in most areas, decreasing snowpacks (Mote and Sharp 2016). Several studies indicate that the length of the fire season and duration of extreme fire weather also are increasing (Abatzoglou and Kolden 2013; Littell et al. 2009; Westerling et al. 2014). These changes are projected to have significant effects on ecosystem processes, species distributions, and community composition (e.g., Blumenthal et al. 2016; Schlaepfer et al. 2012c). Changing climate conditions can also influence the abundance and spread of insects and diseases and increase the stress levels of host species, making them more susceptible to the effects of insects and diseases and causing higher mortality (IPCC 2014). Developing an understanding of the changes that are occurring is essential for evaluating the effects of ongoing management actions and determining effective adaptation strategies (text box 3.1).

Drought

From a meteorological perspective drought is defined as the accumulated imbalance between the supply of water and the demand for water by plants, animals, the atmosphere, the soil column, and humans (Kunkel et al. 2013a,b). Drought can also be defined from other perspectives including hydrologic (e.g., streamflow), agricultural (e.g., ecosystem productivity), or socioeconomic (Luce et al. 2016). Determining whether a drought is occurring can take a relatively longer time for areas where the effects of drought may accumulate slowly, such as forests and sagebrush ecosystems. Ecological indicators of drought exist for rangelands and can be listed sequentially: Water shortages stress plants and animals, vegetation production is reduced, plant mortality increases, plant cover is reduced, amount of bare ground increases, soil erosion becomes more

Text Box 3.1—Monitoring Climate Change Effects

Long-term monitoring results can be used to track changes in species and ecosystems induced by the effects of climate change. At the biome to mid-scale, remote sensing can be used to detect changes in environmental conditions, such as the duration of snowpacks and seasonal soil moisture availability, and the effects on ecosystems, such as changes in plant phenology and productivity. Remote sensing can also be used to monitor changes in persistent ecosystem threats, such as plant invasions and wildfire patterns. Information on the rates and direction of change across the sagebrush biome can be used to prioritize resource allocation for management of invasive species, wildfire and vegetation, and wild horses and burros. It can also be used to determine where to target adaptation strategies to maintain landscape connectivity, ecosystem redundancy, and refugia.

Combining ground-based monitoring with remote sensing can help scale-up results to assess which species and ecosystems may be most vulnerable to climate change. Focusing monitoring efforts on climate transition zones and areas projected to exhibit rapid change (e.g., rapid warming events, loss of snowpack, extreme drought) can provide much needed information on climate change effects. Information on these changes can be used to identify effective adaptation strategies, such as maintaining or enhancing key structural and functional groups, increasing genetic diversity, facilitating community adjustments through species transitions, and planning for and responding to disturbance. Monitoring following changes in management or after treatments can be used to verify the effectiveness of management strategies designed to help ecosystems transition to the new climatic conditions.

prevalent, habitat and food resources for wildlife are reduced, wildlife mortality increases, rangeland fires may increase, some insect pests and invasive weeds may increase, forage value and livestock carrying capacity decrease, and then, economic depression in the agricultural sector sets in (Finch et al. 2016).

Drought adaptation measures with shorter-term and longer-term horizons have been identified for rangelands and forests across the western United States (see Briske et al. 2015; Finch et al. 2016; Joyce et al. 2013). Planning for a drought involves developing a drought management plan (UNL-NDMC 2012; examples available at http://drought.unl.edu/ranchplan/WriteaPlan.aspx). Management actions vary regionally and reflect the resources available to cope with drought. In general, the goal is to minimize the risk of environmental degradation and loss of ecosystem function. Planning for adaptation actions is most successful if coordinated across all land ownerships and if management plans consider the next drought as well as the current drought and its aftermath (Finch et al. 2016).

Current management actions may need to be reexamined with the onset of drought. For example, adaptation actions with respect to livestock management during the drought include: reducing stocking rate to allow plant recovery; using fencing and other developments to manage livestock distribution; using drought-resistant restoration species; using drought-adapted stock; adjusting season of use; implementing a deferred grazing system; developing, restoring, or reclaiming water sources; providing shade structures for livestock; reducing the time livestock graze a specific grazing unit; increasing the time between periods of grazing (rest); and testing new techniques for responding to drought.

With respect to restoration, climate and weather models are now available that can be used to help inform the timing of planting (Hardegree et al. 2012). Under certain conditions, it may be beneficial to delay planting and shift the focus to restoring areas with less desirable species. For example, implementing measures to control crested wheatgrass (*Agropyron cristatum*) during dry years and seeding native grass in wetter years may result in more effective restoration in the West-Central Semiarid Prairies (Bakker et al. 2003). To mitigate the longer-term impact of drought or other abiotic stressors, plant material selection should consider the adaptive capacity of different species and genetic variation within species (Richardson et al. 2012). Assisted migration can be considered for areas where high rates of climate change are expected and the likelihood of success has been evaluated (table 3.1). These decisions will be critical given the potential for increased frequency and duration of drought in the future.

Snowpack and Dust

Total snowfall has been declining precipitously in the West since the 1920s (Kunkel et al. 2009). Maximum seasonal snow depth declined from winter 1960–1961 to winter 2014–2015 across North America, and other studies showed declines in snow cover as well (Kunkel et al. 2016). A recent analysis of April snowpack data, which are used extensively for spring streamflow forecasting, indicated declines at more than 90 percent of the sites when measured from 1955 to 2016 (Mote and Sharp 2016). The average change across all sites amounted to about a 23-percent decline in snow water equivalent. These decreases were observed throughout the western United States, with the most prominent declines in Washington, Oregon, and the northern Rockies (Mote and Sharp 2016).

Decreases in snowpack may not affect overall patterns of soil water availability if precipitation that arrives during the cold season simply switches from snow to rain (Schlaepfer et al. 2012a). However, increases in soil temperature and associated decreases in soil water availability due to longer growing seasons and higher evapotranspiration may influence plant species establishment and survival and thus community composition (Palmquist et al. 2016a,b).

Drought, wildfire, and agricultural activities in the western United States contribute to dust in the atmosphere, which settles on snow-covered areas in the winter. Over the last decade, the number of dust-on-snow events increased in the Colorado Rocky Mountains (Painter et al. 2007; Toepfer et al. 2006). Dust-on-snow events reduce duration of snow cover (Painter et al. 2007), increase rate of snowmelt associated with more extreme dust deposition, and produce earlier peak stream flows of 1 to 3 weeks (Livneh et al. 2015; Painter et al. 2012; Steltzer et al. 2009). As a result of these dust-on-snow events, snow chemistry increases in pH, calcium content, and acid neutralizing capacity with more pronounced effects at upper elevations than lower elevation forested sites (Rhoades et al. 2010).

Effects of decreasing snowpack on sagebrush ecosystems will be widespread, but are likely to be most significant in areas with measurable changes in the amount and duration of snowpack. The most vulnerable areas are likely to be those that previously retained snow cover for all or most of the winter, or where winter snowpack was critical to recharge deep soil water. Adaptation strategies specific to these areas have not been developed (but see David 2013). However, identifying these areas and managing them to sustain ecological functions and reduce the impact of existing ecological, land use, and development stressors can facilitate adaptation (table 3.1). Monitoring these areas for changes in soil moisture and temperature and in species composition can provide information on (1) establishment and spread of nonnative invasive plant species and the need for intervention and (2) the need for community adjustments through species transitions.

Fire Regimes

Higher temperatures associated with climate change have been linked to increases in fire size and longer fire seasons and durations of extreme fire weather in forested ecosystems (Westerling et al. 2014). Although some have suggested that these relationships also exist for the western portion of the sagebrush biome, recent analyses of LANDFIRE data (1984–2014) for the Basin and Range, Snake River Plain, and Columbia Plateau ecoregions (fig. 1.1) fail to show significant changes in number of large fires per year, total fire area per year, or 90th percentile large fire size per year. However, these analyses do point toward increasing total fire area and 90th percentile fire size (Dennison et al. 2014). In addition, analyses of fire patterns in juniper and piñon land cover types show that the fire season started earlier and ended later in the Basin and Range ecoregions over the same 30-year study period (1984–2014) (Board et al. 2018).

Both temperature and amount and seasonality of precipitation influence fire regimes. In the Basin and Range, Snake River Plain, and Columbia Plateau ecoregions most precipitation arrives as winter snow and rain, and woody species, such as sagebrush, tend to dominate vegetation communities (Part 1, section 4). In these areas, most fires burn in July and August. Fire intensities are typically moderate to high and extreme fire weather can result in extensive fire spread (Brown 1982; Romme et al. 2009). In contrast, the Northwestern Plains and portions of the Wyoming Basin and Southern Rockies receive more summer precipitation and most fires burn earlier in the year. These areas have higher relative abundance of grasses and usually exhibit moderate fire spread (Brown 1982; Romme et al. 2009). Fire regimes are further influenced by fire season length, which varies from about 90 days per year in cooler and moister ecoregions to more than 135 days per year in warmer and drier ecoregions (Board et al. 2018). Changes in amount and seasonality of precipitation may cause shifts in relative abundances of woody species and grasses and thus live fuel moisture dynamics, which will affect fire behavior. Further increases in temperature without compensating increases in precipitation, especially during the growing season, will continue to cause greater aridity, longer fire seasons, and more extreme fire weather across much of the sagebrush biome (Dai 2013).

Changes in precipitation due to climate change may have very different effects than changes in temperature on the locations and characteristics of wildfires in sagebrush ecosystems, and these effects are likely to differ within and across ecoregions. The relationships between precipitation and fire exhibit high regional variability due to the heterogeneity of topography, climate, soils, vegetation, and land use (Littell et al. 2009; Pilliod et al. 2017). In general, warmer and drier areas characterized by Wyoming big sagebrush (*Artemisia tridentata* ssp. *wyomingensis*) at lower elevations have the potential for large fires to burn every summer, but are **fuel limited** and do not always have enough fuel to burn (Abatzoglou and Kolden 2013; Littell et al. 2009; Westerling et al. 2014). These areas often require 1 or more years with above-normal precipitation to create sufficient fuel for large wildfires (Crimmins and Comrie 2004; Littell et al. 2009; Pilliod et al. 2017; Westerling et al. 2014). At higher elevations, temperatures become cooler, precipitation usually increases, and ecosystems become increasingly **energy limited** in that they have enough fuel to support fires every summer, but may not be dry enough to burn (Abatzoglou and Kolden 2013; Littell et al. 2009; Westerling et al. 2014). Mountain big sagebrush (*Artemisia tridentata* ssp. *vaseyana*) and mountain shrub communities exhibit these characteristics. These areas often require warmer and drier conditions to decrease fuel moisture sufficiently for large wildfires to burn.

Invasive annual grasses are influencing both the areas burned and fire size through the invasive annual grass-fire cycle, primarily in relatively warm and dry areas, where most precipitation arrives in winter and spring (Balch et al.

2013; Pilliod et al. 2017). These grasses increase fine fuels and fuel continuity and thus fire frequency and extent (Balch et al. 2013; Brooks et al. 2004). A 1- to 3-year lag effect of precipitation on both area burned and number of fires in landscapes dominated by cheatgrass (*Bromus tectorum*) is typical (Pilliod et al. 2017). Changes in fire regimes due to invasive annual grasses are most evident in the Snake River Plain and Northern Great Basin (Balch et al. 2013; Pilliod et al. 2017), but these species are projected to expand northward and upwards in elevation with climate warming (Bradley et al. 2016) and are increasing in the eastern portion of the range (Knight et al. 2014; Lauenroth et al. 2014).

Wildfire and vegetation management plays a key role in enhancing resilience to disturbance and resistance to invasive annual grasses in the face of climate change (tables 1.3, 1.4). Primary objectives are to reduce ecosystem vulnerability, increase the capacity of ecosystems to adapt to climate change and its effects, and facilitate species and plant community transitions in response to changing environmental conditions. This entails: (1) reducing fuel loads and continuity to decrease fire severity or extent, or both; (2) lowering competitive suppression of perennial herbaceous species, which largely determine resilience to wildfire and resistance to invasion; and (3) using postfire revegetation to design vegetation communities that maintain higher live fuel moisture and have lower fuel bed continuity and packing ratios (a measure of fuel bed compactness or the fraction of fuel bed volume that is occupied by fuel).

Fuel management to reduce fuel loads and continuity focuses on areas with increased woody fuels (sagebrush or juniper [*Juniperus* spp.] and piñon [*Pinus* spp.]) or fine fuels (grasses and forbs), or both. Woody fuel loading and fine fuel loading interact with fire weather to influence the propensity for wildfires, and decreases in fuel loads can lower the likelihood of wildfires over a range of fire weather conditions (fig. 3.2). A variety of treatments exist to reduce woody fuels, including sagebrush mowing; juniper and piñon cutting, shredding, and mastication; and prescribed fire (table 1.4 and section 4). Similarly, treatments exist to reduce fine fuels, such as herbicide applications and appropriately timed livestock grazing in areas dominated by cheatgrass (Strand et al. 2014; table 1.4 and section 5, this volume). The use of fuel breaks in carefully targeted locations can aid fire suppression efforts (Maestas et al. 2016). For treatments to maintain or increase resilience to wildfire as the climate changes, it is necessary to ensure that sufficient perennial herbaceous species exist before treatment to promote ecosystem recovery and that treatments do not introduce or lower resistance to invasive plants (Chambers et al. 2014a,b). Use of traditional phenological knowledge from Native Americans regarding the appropriate timing of treatments, including use of prescribed fire, shows promise for achieving desired conditions (Armatas et al. 2016; Huffman 2013).

Managing for fuel beds with high temporal and spatial variability could increase resilience to wildfire (Abatzoglou and Kolden 2013; Kay 1995; Littell et al. 2009). This could include treatments that increase sagebrush patch size and variability in gap size (the distances between shrubs and grasses) (Kay 1995). Patch burning to increase vegetation heterogeneity is increasingly used in the U.S. Great Plains, southern Africa, and Australia (e.g., Bird et al. 2013; Brockett et al. 2001; Fuhlendorf et al. 2017; Ricketts and Sandercock 2016; Voleti et al. 2014). It may be possible to create fuel bed heterogeneity in sagebrush ecosystems by conducting patch-scale burns in early spring or late fall to remove conifers and shrubs in ecosystems with moderate to high resilience (e.g., Davies et al. 2008; Pyle and Crawford 1996; Trauernicht et al. 2015).

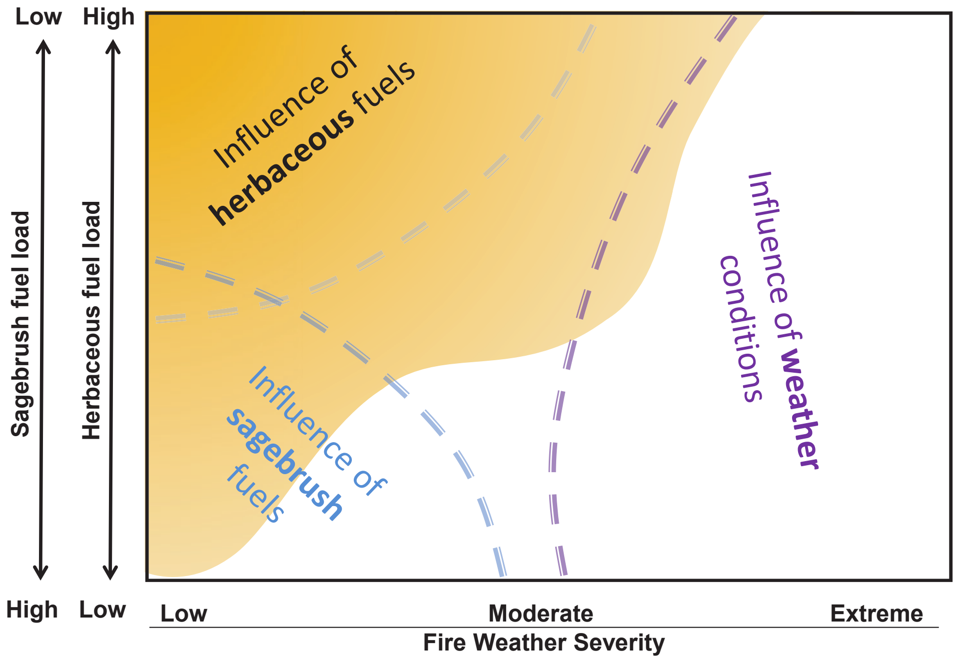

Figure 3.2—The interaction of herbaceous and sagebrush fuels with fire weather severity. In this conceptual model, fuel composition is displayed on the y-axis and fire weather condition is displayed on the x-axis. Low fire weather severity is characterized by high fuel moistures, high relative humidity, low temperature, and low wind speeds, while extreme fire weather is characterized by the opposite conditions. As woody fuel loading or fine fuel loading, or both, increases, fuel packing ratios become more optimal, fuel continuity increases, and less severe fire weather is required for large wildfires. Annual grasses fill interspaces between native fuels (shrubs and bunchgrasses) and are particularly problematic. However, progressive increases in sagebrush or juniper and piñon stand density also lower the severity of fire weather required for large wildfires. Reductions in fuel loads can decrease the likelihood of large wildfires over a range of fire weather conditions. However, extreme fire weather conditions, which are projected to increase in the future, can override the influence of fuel loads and continuity (figure modified from Strand et al. 2014).

Post-wildfire revegetation provides an opportunity to establish vegetation communities with high fuel bed heterogeneity that may be more resilient to wildfire. Resilience to wildfire could be increased by restoring or maintaining plant communities that maintain higher live fuel moisture during dry periods or drought through differences in the relative proportions of herbaceous vegetation to shrubs, varying phenologies and water use patterns, and differences in the cure rate of grasses and forbs (Kay 1995; Palmquist et al. 2016a,b; Schlaepfer et al. 2012b). Also, fuel bed continuity and packing ratio could be decreased by seeding native plant species with growth forms and structures (e.g., size of stems, distance between stems) that are not conducive to carrying fire, even when cured. Most native forbs and some rhizomatous grasses, such as western wheatgrass (*Pascopyrum smithii*), have these properties.

Monitoring the responses of sagebrush ecosystems to wildfire as the climate changes can help inform adaptive management strategies (text box 3.1). At broad scales monitoring changes in wildfire patterns in relation to habitats of species at risk and other resource values can help prioritize resource allocation for preparedness, prevention, suppression, and postfire rehabilitation. At mid- to local scales, information on changes in wildfire area burned and size for specific ecological types or ecological sites that characterize ecoregions provides the basis for adjusting preparedness, prevention, and suppression management strategies over time (section 4). Large changes in species composition and decreased resistance to invasive plants, particularly invasive annual grasses, indicate decreased resilience to wildfire and the need to modify postwildfire rehabilitation strategies.

Changes in Species Distributions and Community Composition

The changes in precipitation and temperature regimes occurring as a result of climate warming are projected to have large consequences for species distributions and, because individual species differ in their climatic requirements, for community composition (Part 1, section 5.2). The distribution of species such as big sagebrush is projected to move to the north and upward in elevation (Bradley 2010; Homer et al. 2015; Schlaepfer et al. 2012c; Still and Richardson 2015). For juniper and piñon species, habitat with suitable climate is projected to move north and upslope with principal gains in Colorado and southwest Wyoming and losses in the Southwest (Rehfeldt et al. 2006, 2012). Cheatgrass is likely to spread upward in elevation while red brome (*Bromus rubens*) moves northward or increases its abundance in the Cold Deserts and Colorado Plateau, or both (Bradley et al. 2016). Decreases in average summer precipitation or prolonged summer droughts could enable cheatgrass invasion into sagebrush ecosystems that are currently more resistant to invasion and resilient to fire disturbance (Bradley et al. 2016; Mealor et al. 2013), such as the northern mixed-grass prairie, allowing it to more successfully colonize what is currently considered a largely invasion-resistant grassland (Blumenthal et al. 2016).

Climate adaptation strategies for the sagebrush biome are designed to facilitate adaptation of species and communities to a warming climate and to reduce the risk of nonnative invasive plant species introduction, establishment, and spread. An understanding of the rates and magnitude of projected change (see Part 1, Appendix 3) can help managers to prioritize areas for different types of management actions (table 3.1). Areas that are likely to support big sagebrush ecosystems in the future may be good candidates for proactive weed and fire management. Areas that may become more suitable for big sagebrush over time may be candidates for assisted migration during restoration activities. Areas that are unlikely to support big sagebrush ecosystems in the future require careful evaluation to determine the types of ecosystems they are likely to support and whether they merit investment in conservation and restoration resources. Careful assessment of connectivity requirements for sagebrush-dependent species to survive and persist as the climate changes can help inform decisions about where to place limited conservation and restoration resources (Part 1, Appendix 9).

Successful adaptation will include monitoring along climate transition zones to detect changes in both soil temperature and moisture regimes and species composition. Consideration of scale will ensure that planning at broad scales promotes strategies such as landscape connectivity, ecosystem redundancy, and refugia, and that planning at more local scales promotes strategies such as maintaining or enhancing key structural and functional groups, increasing genetic diversity, facilitating community adjustments through species transitions, and planning for and responding to disturbance.

Insects and Disease

Major insect pests and diseases affecting plant and sagebrush dependent wildlife species are poorly identified and studied in sagebrush ecosystems. For example, Aroga moth (*Aroga websteri*), or sagebrush defoliator, is a native moth that experiences periodic outbreaks over large areas affecting sagebrush and sage-grouse habitat quality and quantity. West Nile virus (*Flavivirus* spp.) is a recently established disease in the western hemisphere with potential to greatly reduce many avian species populations such as GRSG.

Outbreaks of the native Aroga moth can damage and kill sagebrush over local to mid-scales, although the only documented outbreaks to date have been in the Cold Deserts in the western part of the sagebrush biome. Anecdotal evidence from the northern Great Basin indicates that Aroga moth outbreaks can be associated with years that have much larger than average fires (Tony Svejcar, retired Rangeland Scientist and Research Leader, Burns, OR, personal communication, 2012). Outbreaks are associated with warm conditions from mid-May through mid-June, during the first and second instar development, followed by high precipitation in June and July, during the fourth and fifth instar development (Bolshakova 2013; Bolshakova and Evans 2016). Peak larval abundance occurs around 239 degree-days (accumulated since January 1 using a base temperature of 5 °C [41 °F]), so managers can track degree-days and monitor larval populations to determine when an outbreak is possible (Bolshakova and Evans 2016). How changes in climate may alter the likelihood of such outbreaks is unclear. Outbreaks may occur at the same frequency but earlier in the year as conditions warm, or the frequency may decline due to the combination of warming temperatures and changes in precipitation timing.

Higher moth survival and abundance are also associated with northerly aspects at mid-elevation, suggesting that sagebrush canopy cover may play an as-yet poorly understood role in outbreaks (Bolshakova and Evans 2014). These sites typically experience lower daily and annual temperature fluctuation, greater snow accumulation, and slower snowmelt, thereby creating more favorable conditions for moth larvae and adults (Bolshakova and Evans 2014). More homogeneous stands of sagebrush may serve as epicenters for outbreaks (Bolshakova 2013; Bolshakova and Evans 2014), suggesting that enhancing heterogeneity of sagebrush cover may limit the size and impact of future outbreaks.

Sage-grouse mortality from West Nile virus typically occurs between mid-May and mid-September with peak mortality in July and August (Walker and Naugle 2011), which are also the warmest and driest months. Sage-grouse frequently use ponds, springs, and other standing water sources during hot weather, which are the same sites used by *Culex tarsalis*, the primary mosquito species that transmits West Nile virus to birds (Schrag et al. 2010; Walker and Naugle 2011). Increasing storm intensity that results in more runoff than infiltration, and the potential need to develop additional water sources for domestic and wild ungulates or for irrigation, could result in creating new or enhancing existing breeding sites for *C. tarsalis* mosquitoes. Where West Nile virus is present, fencing or other modifications to watering sites to limit trampling by livestock, wild horses and burros, and wild ungulates can reduce the number of potential *Culex* mosquito breeding sites (NTT 2011, p. 61). Ponds and tanks can be constructed or modified to discourage breeding mosquitoes (Doherty 2007; Walker and Naugle 2011).

Greenhouse Gas Emissions and Carbon Storage

Actions taken to maintain or enhance the resilience of sagebrush ecosystems to disturbance have implications for greenhouse gas emissions and carbon storage. Semiarid ecosystems strongly influence the trend and interannual variability in the global carbon balance, in part due to widespread woody species expansion and high interannual variability in temperature and precipitation (Ahlström et al. 2015). In wetter years, semiarid systems are typically carbon sinks, while in drier years they tend to be carbon sources because respiration exceeds photosynthesis. In more-or-less average years, semiarid systems tend to be more carbon neutral with uptake by photosynthesis roughly equal to release by respiration (Ahlström et al. 2015; Svejcar et al. 2008).

Actions intended to avoid or halt the spread of invasive annual grasses by increasing resilience to disturbance and resistance to invasion and by restoring invaded sites to sagebrush communities would enhance carbon storage and reduce potential greenhouse gas emissions at all scales. In sagebrush ecosystems most carbon is stored belowground in the roots (Rau et al. 2011a). Conversion of native sagebrush ecosystems to annual grassland converts a greenhouse gas sink into a greenhouse gas source with reductions in aboveground and especially belowground carbon storage (Bradley et al. 2006; Germino et al. 2016; Rau et al. 2011a).

Juniper and piñon expansion and infill in sagebrush ecosystems increase aboveground carbon storage many-fold due to the large increase in biomass, but the impacts belowground are not well understood (Rau et al. 2011b, 2012). Once aboveground tree cover equals 50 percent, resilience to disturbance and resistance to invasive annual grasses drop, and the site may become susceptible to invasive annual grasses after fire (Rau et al. 2012) or other stand-replacing disturbances. The tree cover at which this reduction occurs may be lower on less productive sites.

Further, juniper and piñon expansion and infill reduce total soil nitrogen, which has long-term adverse implications for carbon storage in deep soil, where the carbon pool is very stable (Rau et al. 2012). Juniper and piñon expansion and infill can lengthen fire return intervals but greatly increase the biomass consumed during fire in comparison to sagebrush dominated ecosystems. Consequently, the science is unclear as to the long-term tradeoffs in potential greenhouse gas emissions. Even though the increase in biomass from tree cover would seem more consistent with increasing carbon storage, over the longer term it may be less sustainable than maintaining or restoring sites to sagebrush ecosystems. Short-term greenhouse gas emissions and reductions in carbon storage from projects intended or designed to reduce juniper and piñon expansion and restore sage-grouse habitat are acceptable tradeoffs (CEQ 2016, p. 18). Management objectives to increase carbon storage that are consistent with maintaining habitat and key ecosystem functions will be most beneficial in the long term.

References

Abatzoglou, J.T.; Kolden, C.A. 2013. Relationships between climate and macroscale area burned in the western United States. International Journal of Wildland Fire. 22: 1003–1020.

Ahlström, A.; Raupach, M.R.; Schurgers, G.; [et al.]. 2015. The dominant role of semi-arid ecosystems in the trend and variability of the land CO2 sink. Science. 348: 895–899.

Armatas, C.A.; Venn, T.J.; McBride, B.B.; [et al.]. 2016. Opportunities to utilize traditional phenological knowledge to support adaptive management of social-ecological systems vulnerable to changes in climate and fire regimes. Ecology and Society. 21: 16.

Bainbridge, D.A. 2007. A guide for desert and dryland restoration: New hope for arid lands. Covelo, CA: Island Press. 416 p.

Bakker, J.D.; Wilson, S.D.; Christian, J.M.; [et al.]. 2003. Contingency of grassland restoration on year, site, and competition from introduced grasses. Ecological Applications. 13: 137–153.

Balch, J K.; Bradley, B.A.; D'Antonio, C.M.; [et al.]. 2013. Introduced annual grass increases regional fire activity across the arid western USA (1980–2009). Global Change Biology. 19(1): 173–183.

Bentz, B.J.; Duncan, J.P.; Powell, J.A. 2016. Elevational shifts in thermal suitability for mountain pine beetle population growth in a changing climate. Forestry. 89: 271-283.

Bestelmeyer, B.T.; Tugel, A.J.; Peacock, G.L.J.; [et al.]. 2009. State-and-transition models for heterogeneous landscapes: A strategy for development and application. Rangeland Ecology and Management. 62: 1–15.

Bird, R.B.; Tayor, N.; Codding, B.F.; [et al.]. 2013. Niche construction and dreaming logic: Aboriginal patch mosaic burning and varanid lizards (*Varanus gouldii*) in Australia. Proceedings of the Royal Society B. 280: 20132297.

Blumenthal, D.M.; Kray, J.A.; Ortmans, W.; [et al.]. 2016. Cheatgrass is favored by warming but not CO2 enrichment in a semi-arid grassland. Global Change Biology. 9: 3026–3038.

Board, D.I.; Chambers, J.C.; Miller, R.M.; [et al.]. 2018. Fire patterns in piñon and juniper land cover types in the semiarid Western U.S. from 1984 through 2013. RMRS-GTR-372. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 57 p. https://www.fs.usda.gov/treesearch/pubs/55663. [Accessed May 23, 2018].

Bolshakova, V.L. 2013. Causes and consequences of local variability in *Aroga websteri* Clarke abundance over space and time. Dissertation. Logan, UT: Utah State University. 153 p. https://digitalcommons.usu.edu/etd/2014/. [Accessed May 23, 2018].

Bolshakova, V.L.; Evans, E.W. 2014. Spatial and temporal dynamics of *Aroga* moth (Lepidoptera: Gelechiidae) populations and damage to sagebrush in shrub steppe across varying elevation. Environmental Entomology. 43: 1475–1484.

Bolshakova, V.L.; Evans, E.W. 2016. Pheonology of the sagebrush defoliating moth *Aroga websteri* (Lepidopters:Gelechiidae), with application to population irruptions. Annals of the Entomological Society of America. 109: 424–431.

Bradley, B.A. 2010. Assessing ecosystem threats from global and regional change: Hierarchical modeling of risk to sagebrush ecosystems from climate change, land use and invasive species in Nevada, USA. Ecography. 33: 198–208.

Bradley, B.A.; Curtis, C.A.; Chambers, J.C. 2016. *Bromus* response to climate and projected changes with climate change. In: Germino, M.J.; Chambers, J.C.; Brown, C.S., eds. Exotic brome-grasses in arid and semiarid ecosystems of the western U.S. New York, NY: Springer: 257–274.

Bradley, B.A.; Houghton, R.A.; Mustard, J.F.; [et al.]. 2006. Invasive grass reduces aboveground carbon stocks in shrublands of the western US. Global Change Biology. 12: 1815–1822.

Briske, D.D.; Fuhlendorf, S.D.; Smeins, F.E. 2005. State-and-transition models, thresholds, rangeland health: A synthesis of ecological concepts and perspectives. Rangeland Ecology and Management. 58: 1–10.

Briske, D.D.; Joyce, L.A.; Polley, H.W.; [et al.]. 2015. Climate change adaptation on rangelands: Linking regional exposure with diverse adaptive capacity. Frontiers in Ecology and Environment. 13: 249–256.

Brockett, B.H.; Biggs, H.C.; van Wilgen, B.W. 2001. A patch mosaic burning system for conservation areas in southern Africa savannas. International Journal of Wildland Fire. 10: 169–183.

Brooks, M.L.; D'Antonio, C.M.; Richardson, D.M.; [et al.]. 2004. Effects of invasive alien plants on fire regimes. BioScience. 54(7): 677–688.

Brown, J.K. 1982. Fuel and fire behavior prediction in big sagebrush. Res. Pap. INT-RP-290. Ogden, UT: U.S. Department of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station. 10 p.

Bucharova, A. 2017. Assisted migration within species range ignores biotic interactions and lacks evidence. Restoration Ecology. 25: 14–18.

Butler, P.R.; Swanston, C.W.; Janowiak, M.K.; [et al.]. 2012. Adaptation strategies and approaches. In: Swanston, C.W.; Janowiak, M.K., eds. Forest adaptation resources: Climate change tools and approaches for land managers. Gen. Tech. Rep. NRS-GTR-87. Newtown Square, PA: U.S. Department of Agriculture, Forest Service, Northern Research Station: 15–34.

Carwardine, J.; O'Connor, T.; Legge, S.; [et al.]. 2011. Priority threat management to protect Kimberley wildlife. CSIRO Tech. Rep. Brisbane, Australia: CSIRO Ecosystem Sciences. csiro:EP102445. https://doi.org/10.4225/08/584ee8578e288

Caudle, D.; DiBenedetto, J.; Karl, M.; [et al.]. 2013. Interagency ecological site handbook for rangelands. Washington, DC: U.S. Department of the Interior, Bureau of Land Management; U.S. Department of Agriculture, Forest Service and Natural Resources Conservation Service. http://directives.sc.egov.usda.gov/viewerFS.aspx?hid=33150. [Accessed May 23, 2018].

Chambers, J.C.; Beck, J.L.; Bradford, J.B.; [et al.]. 2017. Science framework for conservation and restoration of the sagebrush biome: Linking the Department of the Interior's Integrated Rangeland Fire Management Strategy to long-term strategic conservation actions. Part 1. Science basis and applications. RMRS-GTR-360. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 213 p.

Chambers, J.C.; Bradley, B.A.; Brown, C.A.; [et al.]. 2014a. Resilience to stress and disturbance, and resistance to *Bromus tectorum* L. invasion in the cold desert shrublands of western North America. Ecosystems. 17: 360–375.

Chambers, J.C.; Miller, R.F.; Board, D.I.; [et al.]. 2014b. Resilience and resistance of sagebrush ecosystems: Implications for state and transition models and management treatments. Rangeland Ecology and Management. 67: 440-454.

Cook, B.I.; Ault, T.R.; Smerdon, J.E. 2015. Unprecedented 21st century drought risk in the American Southwest and Central Plains. Science Advances. 1: e1400082.

Council on Environmental Quality [CEQ]. 2016. Memorandum for heads of federal departments and agencies. Issued Aug. 1, 2016. Washington, DC: Council on Environmental Quality. 34 p.

Crimmins, M.A.; Comrie, A.C. 2004. Interactions between antecedent climate and wildfire variability across southeastern Arizona. International Journal of Wildland Fire. 13(4): 455–466.

Cross, M.S.; McCarthy, P.D.; Garfin, G.; [et al.]. 2013. Accelerating adaptation of natural resource management to address climate change. Conservation Biology. 27: 4–13.

Dai, A. 2013. Increasing drought under global warming in observations and models. Nature Climate Change. 3: 52–58.

David, E. 2013. Innovative snow harvesting technology increases vegetation establishment success in native sagebrush ecosystem restoration. Plant and Soil. 373: 843–856.

Davies, K.W.; Sheley, R.L.; Bates, J.D. 2008. Does fall prescribed burning of *Artemisia tridentata* steppe promote invasion or resistance to invasion after a recovery period? Journal of Arid Environments. 72: 1076–1085.

Dennison, P.E.; Brewer, S.C.; Arnold, J.D.; [et al.]. 2014. Large wildfire trends in the western United States, 1984–2011. Geophysical Research Letters. 41: 2928–2933.

Doherty, M.K. 2007. Mosquito populations in the Powder River Basin, Wyoming: A comparison of natural, agricultural and effluent coal-bed natural gas aquatic habitats. Thesis. Bozeman, MT: Montana State University. https://scholarworks.montana.edu/xmlui/bitstream/handle/1/1184/DohertyM1207.pdf?sequence=1. [Accessed May 23, 2018].

Field, S.A.; Tyre, A.J.; Jonzen, N.; [et al.]. 2004. Minimizing the cost of environmental management decisions by optimizing statistical thresholds. Ecology Letters. 7: 669–675.

Finch, D.M.; Pendleton, R.L.; Reeves, M.C.; [et al.]. 2016. Rangeland drought: Effects, restoration, and adaptation. In: Vose, J.M.; Clark, J.S.; Luce, C.H.; [et al.], eds. Effects of drought on forests and rangelands in the United States: A comprehensive science synthesis. Gen. Tech. Rep. WO-GTR-93b. Washington, DC: U.S. Department of Agriculture, Forest Service, Washington Office: 155–194.

Fuhlendorf, S.D.; Hovick, T.J.; Elmore, R.D.; [et al.]. 2017. A hierarchical perspective to woody plant encroachment for conservation of prairie-chickens. Rangeland Ecology and Management. 70: 9–14.

Germino, M.J.; Belnap, J.; Stark, J.M.; [et al.]. 2016. Ecosystem impacts of exotic annual invaders in the genus *Bromus*. In: Germino, M.J.; Chambers, J.C.; Brown, C.S., eds. Exotic brome-grasses in arid and semiarid ecosystems of the western U.S. New York, NY: Springer: 61–95.

Halofsky, J.E.; Peterson, D.L.; Dante-Wood, S.K; [et al.], eds. 2018a. Climate change vulnerability and adaptation in the Northern Rocky Mountains [Parts 1 and 2]. Gen. Tech. Rep. RMRS GTR-374. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 475 p.

Halofsky, J.E.; Peterson, D.L.; Ho, J.J.; [et al.], eds. 2018b. Climate change vulnerability and adaptation in the Intermountain Region [Parts 1 and 2]. Gen. Tech. Rep. RMRS-GTR-375. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 509 p.

Hardegree, S.P.; Schneider, J.M.; Moffet, C.A. 2012. Weather variability and adaptive management for rangeland restoration. Rangelands. 34: 53–56.

Heller, N.E.; Zavaleta, E.S. 2009. Biodiversity management in the face of climate change: A review of 22 years of recommendations. Biological Conservation. 142: 14–32.

Hobbs, R.J.; Higgs, E.; Harris, J.A. 2009. Novel ecosystems: Implications for conservation and restoration. Trends in Ecology and Evolution. 24: 599–605.

Homer, C.G.; Xian, X.; Aldridge, C.L.; [et al.]. 2015. Forecasting sagebrush ecosystem components and Greater sage-grouse habitat for 2050: Learning from past climate patterns and Landsat imagery to predict the future. Ecological Indicators. 55: 131–145.

Huffman, M.R. 2013. The many elements of traditional fire knowledge: Synthesis, classification, and aids to cross-cultural problem solving in fire-dependent systems around the world. Ecology and Society. 18: 3.

Intergovernmental Panel on Climate Change [IPCC]. 2014. Climate change 2014: Impacts, adaptation, and vulnerability. In: Barros, V.R.; Field, C.B.; Dokken, D.J.; [et al.], eds. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. New York, NY: Cambridge University Press, 688 p.

Joyce, L.A.; Briske, D.D.; Brown, J.R.; [et al.]. 2013. Climate change and North American rangelands: Assessment of mitigation and adaptation strategies. Rangeland and Ecology Management. 66: 512–528.

Kay, C.E. 1995. Aboriginal overkill and native burning: Implications for modern ecosystem management. Western Journal of Applied Forestry. 10: 121–126.

Knick, S.T.; Hanser, S.E.; Miller, R.F.; [et al.]. 2011. Ecological influence and pathways of land use in sagebrush. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 203–251.

Knick, S.T.; Hanser, S.E.; Preston, K.L. 2013. Modeling ecological minimum requirements for distribution of Greater sage-grouse leks: Implications for population connectivity across their western range, U.S.A. Ecology and Evolution. 3: 1539–1551.

Knight, D.H.; Jones, G.P.; Reiners, W.A.; [et al.]. 2014. Mountains and plains: The ecology of Wyoming landscapes. Second edition. New Haven, CT: Yale University Press. 404 p.

Kunkel, K.E.; Palecki, M.; Ensor, L.; [et al.]. 2009. Trends in twentieth-century U.S. snowfall using a quality-controlled dataset. Journal of Climate. 26: 33–44.

Kunkel, K.E.; Robinson, D.A.; Champion, S.; [et al.]. 2016. Trends and extremes in northern hemisphere snow characteristics. Current Climate Change Reports. 2: 65–73.

Kunkel, K.E.; Stevens, L.E.; Stevens, S.E.; [et al.]. 2013a. Regional climate trends and scenarios for the U.S. national climate assessment. Part 4. Climate of the U.S. Great Plains. NOAA Technical Report, NESDIS 142-4. Washington, DC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration. 82 p.

Kunkel, K.E.; Stevens, L.E.; Stevens, S.E.; [et al.]. 2013b. Regional climate trends and scenarios for the U.S. national climate assessment. Part 5. Climate of the Southwest U.S. NOAA Technical Report NESDIS 142-5. Washington, DC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration. 79 p.

Kunkel, K.E.; Stevens, L.E.; Stevens, S.E.; [et al.]. 2013c. Regional climate trends and scenarios for the U.S. National Climate Assessment. Part 6. Climate of the Northwest U.S. NOAA Technical Report NESDIS 142-6. Washington, DC: U.S. Department of Commerce, National Oceanic and Atmospheric Administration. 83 p.

Lauenroth, W.K.; Schlaepfer, D.R.; Bradford, J.B. 2014. Interactions between climate and habitat loss effects on biodiversity: A systematic review and meta-analysis. Ecosystems. 17: 1469–1479.

Lesica, P.; Deluca, T.H. 1996. Long-term harmful effects of crested wheatgrass on Great Plains grassland ecosystems. Journal of Soil and Water Conservation. 51: 408–409.

Littell, J.S.; McKenzie, D.; Peterson, D.L.; [et al.]. 2009. Climate and wildfire area burned in western U.S. ecoprovinces, 1916–2003. Ecological Applications. 19: 1003–1021.

Livneh, B.; Deems, J.S.; Buma, B.; [et al.]. 2015. Catchment response to bark beetle outbreak and dust-on-snow in the Colorado Rocky Mountains. Journal of Hydrology. 523: 196–210.

Luce, C.H.; Pederson, N.; Campbell, J.; [et al.]. 2016. Characterizing drought for forested landscapes and streams. In: Vose, J.M.; Clark, J.S.; Luce, C.H.; [et al.], eds. Effects of drought on forests and rangelands in the United States: A comprehensive science synthesis. Gen. Tech. Rep. WO-GTR-93b. Washington, DC: U.S. Department of Agriculture, Forest Service: 14–48.

Maestas, J.; Pellant, M.; Okeson, L.; [et al.]. 2016. Fuel breaks to reduce large wildfire impacts in sagebrush ecosystems. Plant Materials Tech. Note No. 66. Boise, ID: U.S. Department of Agriculture, Natural Resources Conservation Service. 30 p.

McKenzie, D.; Littell, J.S. 2017. Climate change and the eco-hydrology of fire: Will area burned increase in a warming western USA? Ecological Applications. 27: 26–36.

Mealor, B.A.; Mealor, R.D.; Kelley, W.K.; [et al.]. 2013. Cheatgrass management handbook: Managing an invasive annual grass in the Rocky Mountain Region. Laramie, WY: University of Wyoming; Fort Collins, CO: Colorado State University. 131 p.

Millar, C.I.; Skog, K.E.; McKinley, D.C.; [et al.]. 2012. Chapter 4. Adaptation and mitigation. In: Vose, J.M.; Peterson, D.L.; Patel-Weynand, T., eds. Effects of climatic variability and change on forest ecosystems: A comprehensive science synthesis for the U.S. forest sector. Gen. Tech. Rep. PNW-GTR-870. Portland, OR: U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station.

Millar, C.I; Stephenson, N.L.; Stephens, S.L. 2007. Climate change and forests of the future: Managing in the face of uncertainty. Ecological Applications. 17: 2145–2151.

Mote, P.W.; Sharp, D. 2016. 2016 update to data originally published in: Mote, P.W.; Hamlet, A.F.; Clark, M.P.; [et al.]. 2005. Declining mountain snowpack in Western North America. B. American Meteorological. Society. 86: 39–49.

Painter, T.H.; Barnett, A.P.; Landry, C.C.; [et al.]. 2007. Impact of disturbed desert soils on duration of mountain snow cover. Geophysical Research Letters. 34: L12502.

Painter, T.H.; Skiles, S.M.; Deems, J.S.; [et al.]. 2012. Dust radiative forcing in snow of the Upper Colorado River Basin: 1. A 6 year record of energy balance, radiation, and dust concentrations. Water Resources Research. 48: W07521.

Palmquist, K.A.; Schlaepfer, D.R.; Bradford, J.B.; [et al.]. 2016a. Mid-latitude shrub steppe plant communities: Climate change consequences for soil water resources. Ecology. 97: 2342–2354.

Palmquist, K.A.; Schlaepfer, D.R.; Bradford, J.B.; [et al.]. 2016b. Spatial and ecological variation in dryland ecohydrological responses to climate change: Implications for management. Ecosphere. 7: e01590

Pilliod, D.S.; Welty, J.L.; Arkle, R.S. 2017. Refining the cheatgrass-fire cycle in the Great Basin: Precipitation timing and fine fuel composition predict wildfire trends. Ecology and Evolution. 7: 8126-8151.

Prein, A.F.; Holland, G.J.; Rasmussen, R.M.; [et al.]. 2016. Running dry: The U.S. Southwest's drift into a drier climate state. Geophysical Research. 43: 1272–1279.

Pyke, D.A. 2011. Restoring and rehabilitating sagebrush habitats. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 531–548.

Pyle, W.H.; Crawford, J.A. 1996. Availability of foods of sage grouse chicks following prescribed fire in sagebrush-bitterbrush. Journal of Range Management. 49: 320–324.

Rau, B.M.; Johnson, D.W.; Blank, R.R.; [et al.]. 2011a. Transition from sagebrush steppe to annual grass (*Bromus tectorum*): Influence on belowground carbon and nitrogen. Rangeland Ecology and Management. 64: 139–147.

Rau, B.M.; Johnson, D.W.; Blank, R.R.; [et al.]. 2011b. Woodland expansion's influence on belowground carbon and nitrogen in the Great Basin U.S. Journal of Arid Environments. 75: 827–835.

Rau, B.M.; Tausch, R.; Reiner, A.; [et al.]. 2012. Developing a model framework for predicting effects of woody expansion and fire on ecosystem carbon and nitrogen in a pinyon-juniper woodland. Journal of Arid Environments. 76: 97–104.

Rehfeldt, G.E.; Crookston, N.L.; Enz-Romero, C.M.; [et al.]. 2012. North American vegetation model for land-use planning in a changing climate: A solution to large classification problems. Ecological Applications. 22: 119–141.

Rehfeldt, G.E.; Crookston, N.L.; Warwell, M.; [et al.]. 2006. Empirical analyses of plant-climate relationships for the western United States. Journal of Plant Science. 167: 1123–1150.

Rhoades, C.C.; Elder, K.; Green, E. 2010. The influence of an extensive dust event on snow chemistry in the Southern Rocky Mountains. Arctic, Antarctic, and Alpine Research. 42: 98–105.

Richardson, B.A.; Shaw, N.L.; Pendleton, R.L. 2012. Chapter 4. Plant vulnerabilities and genetic adaptation. In: Finch, D.M., ed. Climate change in grasslands, shrublands, and deserts of the interior American West: A review and needs assessment. Gen. Tech. Rep. RMRS-GTR-285. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 48–59.

Ricketts, A.M.; Sandercock, B.K. 2016. Patch-burn grazing increases habitat heterogeneity and biodiversity of small mammals in managed rangelands. Ecosphere. 7: e01431.

Romme, W.H.; Allen, C.D.; Bailey, J.D.; [et al.]. 2009. Historical and modern disturbance regimes, stand structures, and landscape dynamics in piñon–juniper vegetation of the western United States. Rangeland Ecology and Management. 62: 203–222.

Sage-grouse National Technical Team [NTT]. 2011. A report on national Greater sage-grouse conservation measures. 74 p. https://eplanning.blm.gov/epl-front-office/projects/lup/9153/39961/41912/WySG_Tech-Team-Report-Conservation-Measure_2011.pdf. [Accessed May 22, 2018].

Schlaepfer, D.R.; Lauenroth, W.K.; Bradford, J.B. 2012a. Consequences of declining snow accumulation for water balance of mid-latitude dry regions. Global Change Biology. 18: 1988–1997.

Schlaepfer, D.R.; Lauenroth, W.K.; Bradford, J.B. 2012b. Ecohydrological niche of sagebrush ecosystems. Ecohydrology. 5: 453–466.

Schlaepfer, D.R.; Lauenroth, W.K.; Bradford, J.B. 2012c. Effects of ecohydrological variables on current and future ranges, local suitability patterns, and model accuracy in big sagebrush. Ecography. 35: 374–384.

Schrag, A.; Konrad, S.; Miller, S.; [et al.]. 2010. Climate-change impacts on sagebrush habitat and West Nile virus transmission risk and conservation implications for Greater sage-grouse. GeoJournal. 76: 561–575.

Star, J.; Rowland, E.L.; Black, M.E.; [et al.]. 2016. Supporting adaptation decisions through scenario planning: Enabling the effective use of multiple methods. Climate Risk Management. 13: 88–94.

Steltzer, H.; Landry, C.; Painter, T.H.; [et al.]. 2009. Biological consequences of earlier snowmelt from desert dust deposition in alpine landscapes. Proceedings of the National Academy of Science. 106: 11629–11634.

Still, S.M.; Richardson, B.A. 2015. Projections of contemporary and future climate niche for Wyoming big sagebrush (*Artemisia tridentata* subsp. *wyomingensis*): A guide for restoration. Natural Areas Journal. 35: 30–43.

Strand, E.K.; Launchbaugh, K.L.; Limb, R.; [et al.]. 2014. Livestock grazing effects on fuel loads for wildland fire in sagebrush dominated ecosystems. Journal of Rangeland Applications. 1: 35–57.

Svejcar, T.; Angell, R.; Bradford, J.A.; [et al.]. 2008. Carbon fluxes on North American rangelands. Rangeland Ecology and Management. 61: 465–474.

Swetnam, T.W.; Allen, C.D.; Betancourt, J.L. 1999. Applied historical ecology: Using the past to manage for the future. Ecological Applications. 9: 1189–1206.

Toepfer, S.; Borgeson, L.; Edgerly, B.; [et al.]. 2006. The spatial distribution and impact on avalanche conditions of a dust-on-snow event in the Colorado Rocky Mountains. In: International snow science workshop; 2006 October 1–6; Telluride, CO: 913–916. http://arc.lib.montana.edu/snow-science/objects/issw-2006-913-916.pdf. [Accessed May 22, 2018].

Trauernicht, C.; Brook, B.W.; Murphy, B.P.; [et al.]. 2015. Local and global pyrogeographic evidence that indigenous fire management creates pyrodiversity. Ecology and Evolution. 5: 1908–1918.

University of Nebraska-Lincoln, National Drought Mitigation Center (UNL-NDMC). 2012. Managing drought risk on the ranch: A planning guide for Great Plains ranchers. Lincoln, NE: University of Nebraska, National Drought Mitigation Center. 39 p. http://drought.unl.edu/ranchplan/WriteaPlan.aspx. [Accessed May 22, 2018].

U.S. Department of Agriculture, Forest Service [USDA FS]. 2011. National roadmap for responding to climate change. FS-957b. Washington, DC: U.S. Department of Agriculture, Forest Service. 32 p. http://www.fs.fed.us/climatechange/pdf/Roadmapfinal.pdf. [Accessed May 22, 2018].

U.S. Department of Agriculture, Forest Service [USDA FS]. 2012. Future of America's Forest and Rangelands: Forest Service 2010 Resources Planning Act Assessment. Gen. Tech. Rep. WO-GTR-87. Washington, DC: U.S. Department of Agriculture, Forest Service. 198 p.

U.S. Department of the Interior [USDOI]. 2016. Safeguarding America's lands and waters from invasive species: A national framework for early detection and rapid response. Washington, DC: U.S. Department of the Interior. 55 p. https://www.doi.gov/sites/doi.gov/files/National%20EDRR%20Framework.pdf. [Accessed May 22, 2018].

U.S. Department of the Interior, Fish and Wildlife Service [USDOI FWS]. 2010. Rising to the urgent challenge: Strategic plan for responding to accelerating climate change. Washington, DC: U.S. Department of the Interior, Fish and Wildlife Service. 36 p. https://www.fws.gov/home/climatechange/pdf/CCStrategicPlan.pdf. [Accessed May 22, 2018].

U.S. Department of the Interior, National Park Service [USDOI NPS]. 2013. Using scenarios to explore climate change: A handbook for practitioners. Fort Collins, CO: U.S. Department of the Interior, National Park Service Climate Change Response Program. https://www.nps.gov/subjects/climatechange/upload/CCScenariosHandbookJuly2013.pdf. [Accessed May 22, 2018].

Voleti, R.; Winter, S.L.; Leis, S. 2014. Patch burn-grazing: An annotated bibliography. Papers in Natural Resources 462. 12 p. http://digitalcommons.unl.edu/natrespapers/462. [Accessed May 22, 2018].

Walker, B.L.; Naugle, D.E. 2011. West Nile virus ecology in sagebrush habitat and impacts on Greater sage-grouse populations. In: Knick, S.T.; Connelly, J.W., eds. Greater sage-grouse: Ecology and conservation of a landscape species and its habitats. Studies in Avian Biology 38. Berkeley, CA: University of California Press: 127–142.

Westerling, A.; Brown, T.; Schoennagel, T.; [et al.]. 2014. Briefing: Climate and wildfire in Western U.S. forests. In: Sample, V.A.; Bixler, R.P., eds. Forest conservation and management in the Anthropocene: Conference proceedings. Proceedings RMRS-P-71. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station: 81–102.