Chapter 9
Bromus Response to Climate and Projected Changes with Climate Change

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Abstract A prominent goal of invasive plant management is to prevent or reduce the spread of invasive species into uninvaded landscapes and regions. Monitoring and control efforts often rely on scientific knowledge of suitable habitat for the invasive species. However, rising temperatures and altered precipitation projected with climate change are likely to shift the geographic range of that suitable habitat. Here, we review experimental and modeling studies of climatic limits to exotic annual *Bromus* (*Bromus* hereafter) distribution in the Intermountain West in the context of projections of future climate change. We update empirical models of range shifts to test whether *Bromus rubens* L. (red brome) is likely to expand into ranges that become less suitable for *Bromus tectorum* L. (cheatgrass or downy brome). Warming temperatures are likely to create an advantage for *Bromus* species throughout much of the Intermountain West, potentially enhancing invasion into formerly resistant ecosystems if native species mortality increases with warming and drought. *Bromus rubens* is likely to expand into areas of the Southern Great Basin and Colorado Plateau as warmer winters reduce range constraints caused by cold intolerance. However, a primary limitation to exotic annual *Bromus* invasion and expansion is growing season precipitation. Projections for precipitation change are uncertain, but increased precipitation during periods critical for exotic annual *Bromus* germination and growth is forecast for the Northern Great Basin. Increased *Bromus* reproduction and biomass may exacerbate *B. tectorum* invasion and associated fire risk, especially if coupled with longer fire seasons and more extreme

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fire weather. Managers should anticipate both shifts in the overall distribution of *Bromus* species, as well as changes in relative abundance within its existing range.

Keywords Biogeography • Climate change • Experiments • Range limits • Species distribution model

9.1 Introduction

Both native and invasive plants in the Western USA will be affected by changing climate conditions caused by anthropogenic greenhouse gas emissions. However, different physiological tolerances to temperature and precipitation of individual species (see Brooks et al. 2015) will create opportunities for some species but disadvantages for others. Warmer temperatures are leading to species distribution shifts poleward and upward in elevation—a trend that has been observed for thousands of species globally (e.g., Parmesan and Yohe 2003; Root et al. 2003; Chen et al. 2011). For plants, which are relatively dispersal limited (compared to animals), warmer temperatures tend to lead to an earlier spring and longer growing season (Root et al. 2003; Morisette et al. 2009). A longer growing season may give an advantage to some plant species that are able to initiate growth and reproduce as soon as resources are available, e.g., many annual species and species with high phenotypic plasticity in phenology (Willis et al. 2008). Plants that are able to quickly disperse large numbers of propagules into newly suitable habitat are likely to be favored in a changing climate (Johnston 2011). Exotic annual *Bromus* (*Bromus* hereafter) species have highly plastic responses to resource availability (Hulbert 1955; Monaco et al. 2003; James et al. 2011), high reproductive capacity (Mack and Pyke 1983), and propagules that are readily dispersed, particularly in disturbed areas (Johnston 2011). Consequently, it is likely that climate change will tend to favor species such as *Bromus tectorum* L. (downy brome or cheatgrass) and *Bromus rubens* L. (red brome).

Spatial patterns of *Bromus* response to climate will depend in large part on how climate changes across the Intermountain West. Temperature rise, which is a direct consequence of rising greenhouse gases, is fairly consistently projected by climate models (and therefore has relatively low uncertainty, IPCC 2013). In contrast, precipitation changes are an indirect consequence of rising greenhouse gases, dependent on not only overall temperature changes but also spatial patterns of temperature change and prominent ocean circulation patterns such as the El Niño Southern Oscillation and the Pacific Decadal Oscillation. As a result, precipitation change is less consistently projected by climate models (and therefore has higher uncertainty, IPCC 2013). Thus, species response to temperature alone can be predicted more confidently than species response to temperature and precipitation. Because *Bromus* germination, growth, and reproduction are limited by temperature and water availability, the uncertainty in precipitation makes it difficult to make specific geographic predictions of how distribution and abundance are likely to change. As trends in precipitation change come into better focus, adaptive management practices (e.g., Pyke et al. 2015) will add important flexibility for implementing new control efforts in areas where *Bromus* species are likely to expand. In this chapter, we review projected climate changes for the Western USA and highlight studies of *Bromus* response to specific climate factors to identify how climate changes might influence *B. tectorum* or *B. rubens* expansion and/or contraction. We also evaluate *Bromus* interactions with native ecosystems and feedbacks with a dominant ecosystem driver, fire, under projected climate changes. We conclude with a discussion of management implications and research needs.

9.1.1 Climate Projections for the Western USA

The magnitude of climate change into the future is largely dependent on rates of anthropogenic greenhouse gas emissions, which are highly uncertain (IPCC 2013). The standard scenario presented by the IPCC Fifth Assessment Report is RCP4.5, or a “representative concentration pathway” associated with a scenario where greenhouse gas emissions stabilize at a radiative forcing of 4.5 W/m^2 by 2100. Radiative forcing relates to how much of the sun’s radiation the Earth absorbs vs. emits back to space, with higher values associated with more absorption and greater warming. RCP4.5 is roughly equivalent to an atmospheric CO_2 concentration of 650 ppm by 2100 (van Vuuren et al. 2011). (Preindustrial CO_2 concentration was 280 ppm, and levels as of 2014 were roughly 400 ppm.) This may be an optimistic scenario, as emissions trajectories measured between 1990 and 2009 suggest a pathway closer to RCP6.0 or a CO_2 equivalent of roughly 850 ppm by 2100 (Le Quere et al. 2009). Higher emissions by 2100 increase the amount of projected temperature change.

IPCC temperature projections are modeled based on summer (Jun–Aug) vs. winter (Dec–Feb). Based on the RCP4.5 emissions scenario, summer temperatures (Jun–Aug) are projected to rise throughout the Western USA by 1–3 °C by 2050 and by 2–4 °C by 2100 (IPCC 2013). Winter temperatures (Dec–Feb) are projected to rise by 1–3 °C by 2050 and by 1.5–4 °C by 2100, with larger increases in winter temperature forecast for more northern climates (IPCC 2013). All projected future temperatures are outside the current range of variability of mean seasonal temperatures.

IPCC precipitation projections are modeled based on half-year summer (Apr–Sep) vs. winter (Oct–Mar) precipitation. Based on the RCP4.5 scenario, summer precipitation (Apr–Sep) is projected to decrease in the southwest by 0–20 % in both the 2050 and 2100 scenarios, while increasing from 0 to 10 % in the northwest. In both cases, the model ensemble mean is not significantly different from present-day variability of seasonal precipitation (i.e., both projections encompass zero change; IPCC 2013). However, water stress is expected to increase throughout the west because warming temperatures increase evaporation rates even if precipitation remains constant or increases slightly (Seager et al. 2007). Winter precipitation (Oct–Mar) is projected to remain the same in the southwest and increase by 0–10 % by 2050 and 0–20 % by 2100 in the northwest. However, rising temperatures will lead to more precipitation falling as rain vs. snow during the winter and will likely increase drought stress throughout areas historically dependent on winter snowmelt

(Barnett et al. 2005). Overall soil moisture, which is driven by the combination of precipitation and temperature, is projected to decline significantly throughout the west in all seasons (IPCC 2013).

9.1.2 Climate Effects on Native Ecosystems in the Western USA

The relative dominance of exotic annual *Bromus* species and their effects on native ecosystems will be significantly influenced by climate effects on native species. Projected current and future geographic distribution of biomes in North America created by Rehfeldt et al. (2012) generally indicate the following changes in climate suitable area for the Western USA: (1) expansion to higher latitudes and elevations for cold and warm desert biomes, California evergreen forest and woodland and coastal scrub, Great Basin montane scrub, and Rocky Mountain montane conifer forest; (2) contraction of California chaparral, evergreen forests and woodlands and valley grasslands, and Great Basin conifer woodland and shrub-grassland; and (3) relative stability for Great Plains grasslands. For example, the Mojave and the Great Basin Deserts, including the Colorado Plateau ecoregion, are expected to increase in area by 40 % and 45 % by 2060, respectively, and Great Basin montane scrub is expected to increase by 87 %, while Great Basin conifer woodland and shrub-grassland decline by 21 % and 33 %, respectively. But, even in cases where biomes are projected to expand, range shifts in climatically suitable areas are likely to lead to considerable biome turnover. By 2060, only 65 % of the current Mojave Desert and 67 % of the Great Basin Desert will still have climate conditions that exist within the biome extents today (Rehfeldt et al. 2012).

Specific ecosystems that are currently invaded by *Bromus* species are also likely to be increasingly stressed by climate change. Bioclimate envelope models for *Artemisia tridentata* Nutt. (big sagebrush) and other *Artemisia* species, which *B. tectorum* frequently invades, project large decreases in southern latitudes and lower elevations but relatively small increases in northern latitudes and higher elevations (Bradley 2010; Schlaepfer et al. 2012; Still and Richardson 2015; Fig. 9.1). For *A. tridentata* Nutt. ssp. *wyomingensis* Beetle & Young (Wyoming big sagebrush), which occupies the warmest and driest portions of the species range, a 39 % reduction (66 million hectares) in suitable climate is predicted by mid-century (Still and Richardson 2015). Regions predicted to be most vulnerable to loss of climate suitability include the trailing edge (i.e., the southern periphery of the species), and lower elevation areas of the Columbia and Great Basin. Regions that retain or gain climate suitability include higher elevations in the Great Basin, and the northern Great Plains (Bradley 2010; Schlaepfer et al. 2012).

In the Mojave Desert and Colorado Plateau, regions typically invaded by *B. rubens* and to a lesser degree *B. tectorum*, a dominant native shrub *Coleogyne ramosissima* Torr. (blackbrush) is expected to lose much of its current climatic habitat. While the potential range of *C. ramosissima* expands by an estimated 52 % by 2060, most of this

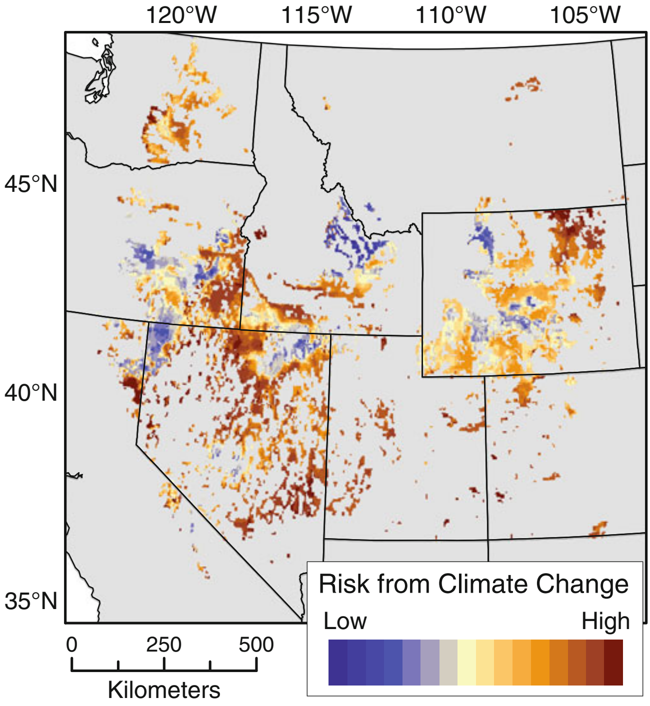

Fig. 9.1 Risk to existing sagebrush (*Artemisia* spp.) populations in the Great Basin associated with climate change. Cooler colors are lower risk, defined as higher likelihood of maintained climatic suitability under multiple general circulation model projections. Warmer colors have high risk from climate change, with few or zero general circulation models projecting future climatic suitability (Adapted from Bradley 2010)

is new range in the Great Basin Desert (Richardson et al. 2014). Climatic habitat loss within the Mojave Desert is nearly 100 % (Richardson et al. 2014). Cold air drainage in certain areas of the topographically diverse Great Basin may result in lower minimum temperatures than generally predicted from climate models and favor cold-adapted ecotypes of *C. ramosissima* (Richardson et al. 2014). Although more studies are needed for native species, case studies for *A. tridentata* and *C. ramosissima* suggest that native ecosystems could be strongly negatively affected by climate change, potentially reducing their resistance to *Bromus* invasion (Chambers et al. 2014a).

In addition to shifting mean climate conditions, greater climate variability likely will favor invasion of annual weeds (Bradley et al. 2010) and negatively affect native species persistence in areas that remain otherwise climatically suitable. Reduced soil moisture availability coupled with greater climate variability can result in reduced resilience or recovery potential of native ecosystems following disturbances such as wildfire (Brooks and Matchett 2003; Brooks and Chambers 2011; Chambers et al. 2014a). In turn, decreased resilience can lower the resistance of these ecosystems to invasive annual grasses like *B. tectorum* and *B. rubens* (Chambers et al. 2014a).

9.2 Climate Effects on *Bromus*

An increased understanding of the effects of climate change on exotic annual *Bromus* is essential for effective management of these species. Changes in the timing and amount of precipitation, and the magnitude of temperature increase, significantly influence climate suitability for individual species. Here we examine the effects of these aspects of climate on exotic annual *Bromus* species with an emphasis on *B. tectorum* and *B. rubens*. We conclude with a discussion of feedbacks among climate, exotic annual *Bromus*, and fire.

9.2.1 Fall/Spring Precipitation

Germination, establishment, and growth of exotic annual *Bromus* are predominantly limited by soil water availability within the range of suitable temperatures for these species (Beatley 1966, 1974; Thill et al. 1979; Roundy et al. 2007). Bradford and Lauenroth (2006) used historical climate data to simulate soil moisture availability over 1000 years within sagebrush steppe ecosystems. They estimated that over the 1000-modeled years, conditions were suitable for *B. tectorum* germination and establishment 72 % of the time in the spring and 25 % of the time in the fall. However, this general finding has large spatial variation. In the Great Basin, the timing of precipitation shifts from primarily winter/spring in the west toward relatively more spring/summer or summer in the east (Chambers et al. 2015). Most of the available information that we review on the life history and physiological traits of *B. tectorum* is from the western portion of the Great Basin, where exotic annual *Bromus* species tend to be more problematic. *B. tectorum* establishment depends on receiving adequate precipitation during periods that are favorable for germination and growth and can occur in fall following precipitation events, and/or in spring when soil moisture increases as a result of snowmelt or spring rains (Mack and Pyke 1984). Survival following fall germination is strongly influenced by short-term drought in the fall (Mack and Pyke 1984). In warmer and drier areas, *B. tectorum* can fail to recruit during low precipitation years, and population growth is constrained by longer-term drought (Meyer et al. 2001; Chambers et al. 2007). Hence, changes in either fall precipitation or winter and spring precipitation could affect germination and establishment of *B. tectorum* across various portions of its range.

High precipitation during the winter/spring in semiarid ecosystems increases the biomass of *Bromus* species (Meyer et al. 2001; Chambers et al. 2007). The accumulation of *B. tectorum* biomass following wet winter/spring conditions can be so dramatic that it can be seen from satellites (Bradley and Mustard 2005). In the Mojave and southern Great Basin, wet winters are associated with strong El Niño events (caused by warm surface waters in the Pacific). Higher *B. tectorum* biomass has been observed for 2 years following El Niño events in areas dominated by the invader. Even if second-year climate conditions are conducive to only average growth, the strong growth response during the second year might be due to prolific seed availability (Bradley and Mustard 2005) and/or higher nutrient availability as a result of more favorable conditions for decomposition during years with wet winter/spring conditions (Jones et al. 2014). El Niño years can greatly increase reproduction and spread of exotic invasive *Bromus* into marginal environments as has been observed for *B. rubens* in the Mojave Desert (Salo 2005) and *B. tectorum* in salt desert ecosystems of the cold desert (Meyer et al. 2001). Conversely, drought during winter/spring can inhibit annual invasive *Bromus* establishment, growth, and reproduction in a variety of ecosystems (Beatley 1966, 1974; Thill et al. 1979; Roundy et al. 2007; Zelikova et al. 2013). Ultimately, the effects of both El Niño years and drought on *Bromus* invasion are strongly moderated by the productivity and species composition of the native community (Chambers et al. 2014a, 2015). Other global change factors that generally promote *Bromus* growth, including rising CO_2 (Smith et al. 1987, 2000; Ziska et al. 2005), increasing N deposition (Hulbert 1955; Adair et al. 2008), and rising temperatures (Thill et al. 1979; Compagnoni and Adler 2014), are all highly dependent on soil water availability, and their effects will largely depend on the magnitude of change in both temperature and precipitation.

While IPCC precipitation projections focus on winter (Oct–Mar) vs. summer (Apr–Sep) precipitation, *Bromus* species are mainly influenced by fall precipitation and/or winter/spring precipitation for germination and growth. The lack of seasonal congruence between projected precipitation changes and the *Bromus* growing season increases uncertainty as to how precipitation change might affect *Bromus*. One consistency in the climate models is a likely increase in winter precipitation in the northern part of the Intermountain West (above ~37° latitude) and an associated increase in anomalously wet winters (Abatzoglou and Kolden 2011; IPCC 2013). This increase could result in expansion of *B. tectorum* into drier areas that are currently marginal habitat but are projected to become wetter at critical times for *Bromus* germination and growth (and, therefore, more suitable). In southern portions of the Intermountain West (below ~37° latitude), winter climate is likely to become drier (Abatzoglou and Kolden 2011; IPCC 2013), which for *B. tectorum* could lead to reduced establishment and growth, lower rates of invasion across much of the area, and potential range contraction where water is more limiting. *Bromus rubens* occurs in areas with slightly lower mean annual precipitation (Brooks et al. 2015) and may be able to expand into some of the areas that are no longer suitable for *B. tectorum* (see Sect. 9.2.5).

9.2.2 Summer Precipitation

The potential range of *Bromus* species depends not only on the life history and physiological traits of *Bromus* but also on the climatic response of potentially invaded ecosystems (Chambers et al. 2014a). *B. tectorum* is established in all 50 US states and almost all Canadian provinces (USDA-NRCS 2014) but typically is not considered invasive and rarely becomes abundant in areas that receive plentiful precipitation in summer (Bradford and Lauenroth 2006; Bradley 2009, 2013). Seasonality of precipitation has important consequences for soil water balance during the year and, thus, plant functional type dominance (Sala et al. 1997) and competitive interactions with *B. tectorum* (Bradford and Lauenroth 2006). The amount of precipitation that is received during the period when temperature, and thus potential evapotranspiration, is low influences the amount of water that is stored in deep soil layers and therefore the relative dominance of woody vs. herbaceous species (Sala et al. 1997). In the west, areas that receive more winter/spring precipitation typically have greater deep soil water storage and are dominated by woody species, which are more effective at using deep soil water. In contrast, areas that receive predominantly summer precipitation tend to be dominated by perennial grasses.

In a regional species distribution modeling study, Bradley (2009) showed that *B. tectorum's* current range within the Great Basin is best explained by summer precipitation. Areas with higher summer precipitation were less likely to be invaded. Increasing summer precipitation may result in less favorable conditions for establishment of *Bromus* and strong competition from native grass species that dominate under this precipitation regime (Bradford and Lauenroth 2006; Bradley et al. 2009). The strength of these competitive interactions likely increases as precipitation increases, and the native community becomes more productive (see Chambers et al. 2015). If average summer moisture availability declines as a result of climate change, Bradley (2009) projected that the land area susceptible to *B. tectorum* (based on climatic suitability) may increase by up to 45 %, particularly in sagebrush steppe in Montana and higher elevation areas of the Colorado Plateau (Box 9.1).

Box 9.1

Figure 9.2 illustrates current and future distributions of *B. tectorum* predicted by Bradley (2009). Here, we used Maxent (Phillips et al. 2006), an implementation of maximum entropy modeling that predicts climatic suitability based on geographic occurrences of a species, to model climatic suitability for *B. tectorum*. Climatic suitability for occurrence implies that the combination of precipitation and temperature conditions at a given location support the germination, growth, and establishment of the species. We used mean temperature in the hottest and coldest months and quarterly precipitation derived from PRISM (averaged for 1951–2006) as our climatic predictors (Daly et al. 2002). Occurrence data were plots surveyed by the Southwest and Northwest Gap Analysis Programs (GAP) (2003–2005) with greater than 25 % reported *B. tectorum* cover. We applied a mask in Maxent to focus only on the surveyed region within the geographic range encompassing all GAP analysis points. We chose a model threshold value that encompassed 90 % of the *B. tectorum* training data with high abundance. This threshold also resulted in the correct identification of 76 % of testing data with *B. tectorum* absence according to the GAP analysis dataset. Loss of average summer precipitation or prolonged summer droughts could enable *B. tectorum* invasion into sagebrush steppe that is currently resistant to invasion and resilient to fire disturbance (Fig. 9.2).

(continued)

Box 9.1 (continued)

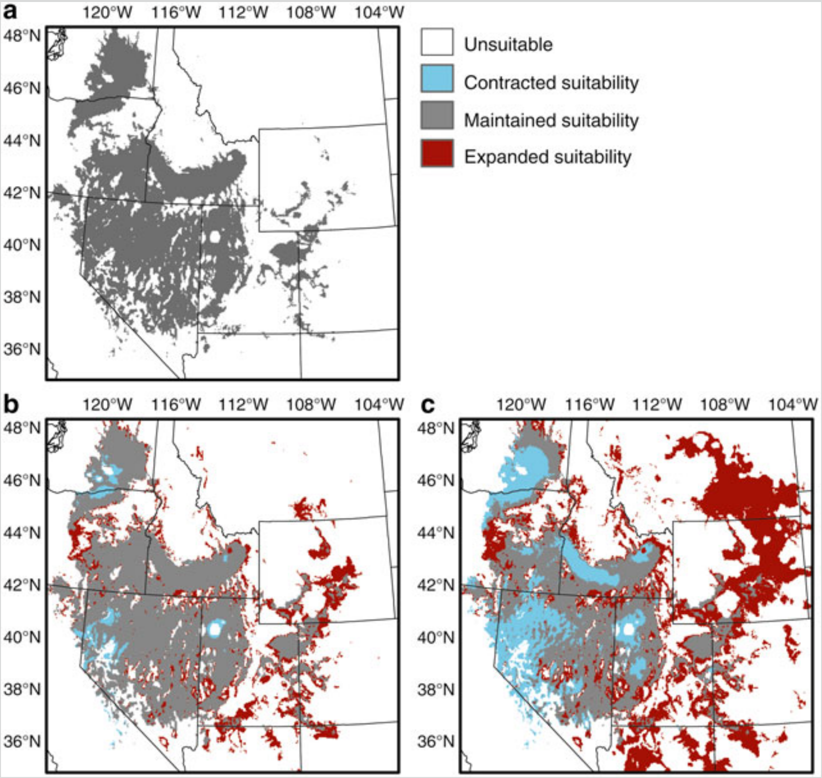

Fig. 9.2 Climatic suitability for high abundance (>25 % cover) of *B. tectorum* in the Intermountain West with varying summer precipitation, specifically (**a**) current, average climate conditions, (**b**) a 25 % loss of average summer precipitation, or (**c**) a 50 % loss of average summer precipitation. *White* remains unsuitable, *light blue* becomes unsuitable, *medium gray* remains suitable, and *dark red* gains suitability for *B. tectorum*

9.2.3 Winter Precipitation

Several studies have tested the response of *B. tectorum* germination and growth to winter snowfall (Griffith and Loik 2010; Concilio et al. 2013; Compagnoni and Adler 2014). At higher elevations, prolonged snowpack delayed *B. tectorum* emergence (Concilio et al. 2013) and reduced seedling survival for individuals that germinated in the fall (Griffith and Loik 2010; Compagnoni and Adler 2014), although these effects did not reduce overall biomass and seed production later in the spring

(Concilio et al. 2013). Colder temperatures in general and colder and wetter winters resulted in lower growth and seed production of *B. tectorum* across elevation gradients in Nevada and Utah (Chambers et al. 2007). Conversely, warmer temperatures increase *B. tectorum* germination rates, survival, growth, and reproduction (Thill et al. 1979; Gasch and Bingham 2006; Zelikova et al. 2013; Compagnoni and Adler 2014). But, the effect of warmer temperatures is only apparent with sufficient water availability (Thill et al. 1979; Zelikova et al. 2013; Compagnoni and Adler 2014). Projected winter warming (IPCC 2013) and loss of snowpack (Mote et al. 2005) at mid elevations are likely to favor *B. tectorum* under non-drought conditions.

9.2.4 Winter and Spring Temperature

Winter and spring temperatures are important predictors of rate and timing of spring germination. Colder temperatures generally decrease germination rates (Thill et al. 1979; Gasch and Bingham 2006; Roundy et al. 2007), plant establishment, and growth and reproduction of *B. tectorum* (Chambers et al. 2007). Spring soil temperature, based on accumulated degree days when soil water availability was sufficient for seed hydration, was the best predictor of the timing and rate of *B. tectorum* germination across elevation gradients in sagebrush ecosystems of Nevada and Utah (Roundy et al. 2007). Plant establishment and reproduction across the same elevation gradients were constrained by low soil temperatures at relatively high elevation (frigid to cryic soils), dependent on growing season conditions at mid elevation (frigid/mesic soils), and optimal under relatively moderate temperature and water availability at lower elevation (xeric/aridic to xeric/mesic soils) (Chambers et al. 2007). At lower elevations with warmer soils (warm-xeric), such as in salt desert ecosystems, constraints shift primarily to precipitation limitation (Meyer et al. 2001; Zelikova et al. 2013). Because *B. tectorum* has a fairly broad temperature tolerance range and high phenotypic plasticity, wide variation in timing of green-up can occur across its range that is dependent on local climates (Hulbert 1955).

In areas where *B. rubens* and *B. tectorum* germinate in the fall, cold winter temperatures could lead to mortality. Both species gain considerable cold temperature tolerance once hardened, so late fall cold snaps before hardening occurs are most likely to kill seedlings (O'Connor et al. 1991; Bykova and Sage 2012). *B. tectorum* has a freezing tolerance of between −30 °C (O'Connor et al. 1991) and −22 °C (Bykova and Sage 2012). *B. rubens* is relatively more susceptible to sudden cold extremes than *B. tectorum* (Bykova and Sage 2012), and cold winter temperatures likely limit *B. rubens*' northern range (Hulbert 1955) and elevational distribution (Salo 2005). Warmer winter temperatures will reduce the likelihood that either *B. tectorum* or *B. rubens* populations that germinate in the fall will experience mortality during the winter over much of their range. *Bromus tectorum* will likely spread upward in elevation, and *B. rubens* might expand northward and/or increase its abundance in the Great Basin Desert and Colorado Plateau (Curtis and Bradley 2015).

9.2.5 Future Range Shifts with Climate Change

Previous distribution modeling studies for *B.tectorum* have suggested that climate change could create large-scale restoration opportunities for invaded areas as climate conditions become unsuitable for the invasive annual grass (Bradley 2009; Bradley et al. 2009). Geographically, these opportunities are most likely at the hotter and drier edge of the species' range, although *B. tectorum* shows local adaptation to warmer range margins (Leger et al. 2009), which may enable persistence for longer than anticipated by climatic suitability models (e.g., Bradley 2009). However, *B. rubens* dominates under hotter and drier conditions characteristic of the Mojave and Sonoran Deserts and may well expand to fill any range that *B. tectorum* vacates. Indeed, while *B. tectorum* is currently more abundant across the southern Great Basin, *B. rubens* is already present at low levels of abundance at lower elevations across much of the Great Basin (Salo 2005) and may be able to spread if *B. tectorum* declines. Hence, rapid *B. rubens* invasion could be possible with altered climate conditions (Box 9.2).

Box 9.2

Figure 9.3 compares climatic suitability models for *B. tectorum* and *B. rubens* under current and future climate changes. Distribution data included over 1,900 locations with abundant (>25 %) *B. tectorum* cover recorded on GAP analysis plots and over 12,000 occurrences of *B. rubens* from a combination of surveys conducted during the 2000s (Southwest Exotic Plant Mapping Program, Nevada Natural Heritage Program, California Invasive Plant Council, Southwest Environmental Information Network, surveys by managers from the Mojave Network Parks). Distribution data for *B. tectorum* were locations with greater than 25 % cover, but points for *B. rubens* were all available occurrences because cover was not collected in any of the compiled survey data. We used Maxent (Phillips et al. 2006) to model current climatic suitability based on the temperature and precipitation variables from PRISM described above. For future conditions, we used an ensemble of average climate projections from 2040 to 2069 obtained from Climate Wizard (Girvetz et al. 2009) for the following climate models: CGCM 3.1-T47, CSIRO MK3.0, ECHAM/MPI-OM, GFDL CM2.1, IPSL CM4, and UKMO HadCM3. Model projections were based on the A2 emissions scenario from the IPCC Fourth

(continued)

Box 9.2 (continued)

Assessment Report, which is consistent with current emissions trajectories (Le Quere et al. 2009). We chose a threshold suitability value that encompassed 90 % of *B. tectorum* or *B. rubens* points and applied the same threshold to each future climate projection. Projections of future climatic suitability based on the climate model scenarios were combined to create a single model in which any pixel predicted to have suitable climate conditions by at least one climate model is considered potential habitat.

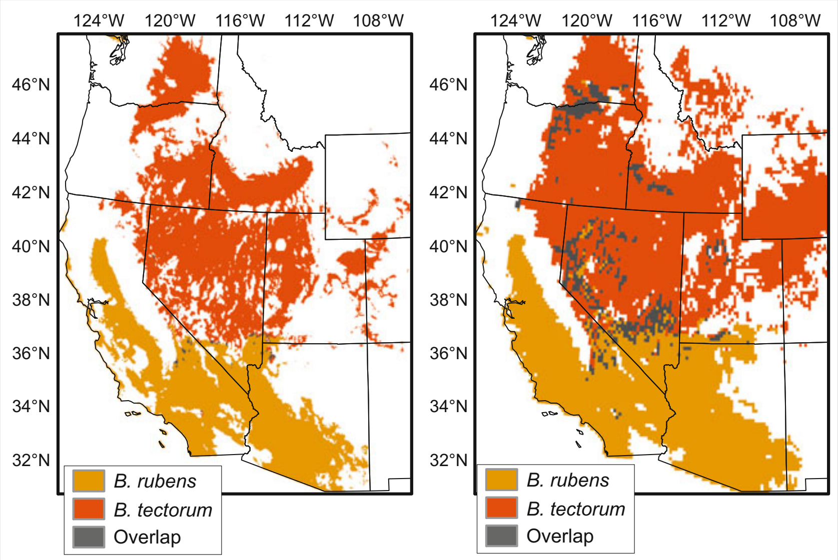

Fig. 9.3 Climatic suitability for *B. tectorum* and *B. rubens* under current (*left*) and future (*right*) climate conditions. *B. rubens* could expand northward into the Great Basin and Colorado Plateau with climate change, potentially negating any restoration opportunities in areas currently invaded by *B. tectorum*. Note: current climate conditions are interpolated to a 4 km spatial resolution, while future climate conditions are scaled to a 12 km spatial resolution

As illustrated by Box 2, climate change could result in a substantial expansion of *B. rubens* northward into the southern Great Basin Desert and Colorado Plateau. The potential area of overlap between the two species also expands considerably, from 2700 km^2 to 86,000 km^2, primarily in the drier portions of the Great Basin Desert. Based on this analysis, it appears unlikely that any restoration opportunities afforded by climate change will last for long. If *B. tectorum* retreats, *B. rubens* may occupy those vacated lands given sufficient available water.

9.2.6 Climate-Fire Feedbacks with Growing Season Precipitation

Invasion of *B. tectorum* and *B. rubens* in semiarid ecosystems greatly increases the amount and continuity of fine fuels and can result in more frequent and larger fires (Brooks 1999; Brooks et al. 2004, 2015). In uninvaded arid and semiarid shrublands where productivity is low, lack of continuous cover decreases the ability of fire to carry from plant to plant (Brooks 1999). The Mojave Desert and lower elevation shrubland in the cold desert historically experienced low fire frequencies. For example, charcoal sediments in low elevation *A. tridentata* shrubland suggest that on average fires burned every 1000 years (Mensing et al. 2006). Estimates from Balch et al. (2013) found similar fire frequencies in sagebrush ecosystems by using remotely sensed data to measure more recent fire frequency. They calculated fire return intervals (i.e., the length of time between repeated burning of the same location) for sagebrush across the Great Basin of between 100 and 300 years. Fires in *B. tectorum*-invaded ecosystems were larger and more frequent, with fire return intervals averaging between 50 and 80 years regionally (Balch et al. 2013) and even more frequently in highly invaded landscapes (Whisenant 1990).

Fires in *B. tectorum*- and *B. rubens*-invaded landscapes are strongly linked to climate and short-term weather conditions and, therefore, are likely to be influenced by climate change. In forested ecosystems, hot and dry weather during the fire season leads to drying of fuels and increased fire frequency (Westerling et al. 2006; Littell et al. 2009). But in low productivity shrubland and desert, fire activity is influenced more by the production of fine fuels, and significant buildup of annual grass and fine fuels occurs in years with wet winters and springs. As a result, the best predictor of fire size and frequency in low productivity shrubland may be wet winters in the year or two prior to the fire (Knapp 1998; Littell et al. 2009; Abatzoglou and Kolden 2013). Similarly, Balch et al. (2013) found that the frequency and size of fires on *B. tectorum*-dominated land cover were correlated with precipitation during the preceding winter. These findings suggest that wet winter climatic conditions at lower elevations where *B. tectorum* is most problematic promote *Bromus* biomass production, increase fine fuels, and will also promote fires. Fires, in turn, result in mortality of many of the dominant native shrub species and an increase in resource availability that can result in increased growth and reproduction of *Bromus* and further enable its expansion (Chambers et al. 2014a), especially in drier regions where shrub recovery is less likely to recover (Taylor et al. 2014).

Climate changes that bring wetter winters to highly invaded regions, as predicted for the Northern Great Basin, are therefore likely to strengthen the fire-*Bromus* feedback, especially under relatively warm conditions. Consistently higher *Bromus* biomass could lead to more frequent and larger fires, facilitating the spread of these exotic invasive grasses. More frequent *Bromus*-driven fires in the northern Intermountain West are also likely if wet winters are accompanied by warmer summers with reduced precipitation, as forecasted with climate change (Diffenbaugh et al. 2008; Abatzoglou and Kolden 2011; IPCC 2013). Conversely, drier winters, or less frequent anomalously wet years in the southwest, could weaken the fire-Bromus feedback.

9.3 Management Implications

The combination of observations, experimental, and modeling studies is increasing our scientific knowledge of *Bromus* species and native community response to climate. However, uncertainty in climate projections coupled with the heterogeneous landscapes of the Intermountain West makes location-specific forecasts a challenge. Management decisions must embrace multiple possible future climate pathways and rely on adaptive management to adjust responses appropriately. Part of *Bromus* response to climate change will depend on the species physiology and life history traits directly, and part will depend on the response of native competitors. Loss of native vegetation increases available resources for *Bromus* species' growth (Roundy et al. 2007; Prevéy et al. 2010) and enables faster dispersal (Johnston 2011). Hence, managing for native competitors and reducing stress and disturbance could limit *Bromus* dispersal and establishment.

Vegetation changes in response to climate are most likely to occur first at the margins of their distribution. For example, *Bromus* species may become more competitive, and therefore more abundant, at higher elevations and latitudes as temperatures warm, while native species may become less competitive and more susceptible to climate extremes at their lower elevation and latitudinal ranges. Monitoring mortality of native perennial species as well as cover of *Bromus* at range margins may give advanced warning of ongoing range shifts and enable adaptive management.

Resilient native ecosystems are important for increasing resistance to *Bromus* invasion (Chambers et al. 2014a). Management actions that promote native species diversity and abundance under changing climate conditions could reduce invasion rates. But historical rates of native plant migration have been estimated to be only 10–30 km per century (McLachlan et al. 2005; Yansa 2006), and many native species may not be able to expand into newly suitable climate within the short period of time in which climate changes are likely to occur (ca. 40–50 years). If native species are unable to colonize newly suitable areas, assisted migration is one possible solution (McLachlan et al. 2007; Richardson et al. 2009; Vitt et al. 2010). Assisted migration can be defined as 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, as a direct management response to climate change (Havens et al. 2015). Plant sources adapted to the new areas would need to be used to ensure successful assisted migration. Also, soil conditions and other environmental characteristics would need to be suitable to the new species (Richardson et al. 2014).

9.4 Research Needs

While a number of studies have focused on the physiological tolerance of *Bromus* species (primarily *B. tectorum*) to climate conditions, fewer studies have considered climate effects on interactions of *Bromus* with native species. Changes in native plants' germination, growth, and mortality rates due to climate will alter overall ecosystem resistance to invasion and resilience following drought events and/or fires. Experimental and modeling studies that test biotic interactions across existing environmental gradients (e.g., Brooks et al. 2015) might provide better insight about overall invasion risk with climate change than studies of *Bromus* independently.

Important information about likely species' response to climate change will come from long-term monitoring sites, experimental studies, and observations across environmental gradients. The high interannual variability of climate conditions across the west creates a challenge for forecasting likely species' response. Many short-to-intermediate-term studies report very different responses of *Bromus* species from 1 year to the next due to different precipitation conditions (e.g., West and Yorks 2002; Chambers et al. 2014b), which makes it difficult to infer longer-term patterns. Prioritizing longer-term, multi-year experiments will improve our ability to project future ecological changes. Further, the importance of collecting and analyzing vegetation response data from long-term monitoring sites at university experimental stations, national parks, US Forest Service monitoring sites, and other public lands cannot be overstated, as they will be critical for understanding vegetation response across a broader range of climatic conditions.

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