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Challenges and limitations to native species restoration in the Great Basin, USA

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Abstract The Great Basin of the western USA is an arid region characterized by high spatial and temporal variability. The region experienced high levels of ecological disturbance during the early period of Euro-American settlement, especially from about 1870–1935. The principal plant communities of the Great Basin are sagebrush steppes, dominated by various *Artemisia* shrubs and perennial bunchgrasses that represent the largest rangeland ecosystem in North America. In low to mid-elevation sagebrush communities, exotic annual grasses have displaced native plant species and are associated with a dramatic increase in size and frequency of wildfires. Degradation in this region is driven by processes that cause the loss of mature bunchgrasses, which, when intact, limit exotic annual grass invasion. Historically, large economic investments to restore degraded Great Basin rangeland through establishment of native bunchgrasses, principally utilizing heavily mechanized agronomic approaches, have been met with limited success. A multitude of environmental factors contribute to the lack of restoration success in this region, but seedling mortality from freezing and drought has been identified as a primary demographic limitation to successful bunchgrass establishment. Novel approaches to overcoming limitations to bunchgrass establishment will be required for restoration success. Increased national concern and a near listing of the greater sage-grouse, a steppe-obligate species, to Endangered Species status, has spurred greater regional support and collaboration across a diversity of stakeholder groups such as state and federal land and wildlife management agencies, county planners, and ranchers.

Keywords Great Basin · Restoration · Sage steppe · Catastrophic fire · Cheatgrass · Medusahead · Bunch grasses

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Introduction to the Great Basin, USA

The goal of this paper is to describe the general setting, past restoration practices, and the potential future for restoration in the Great Basin of the USA. It is important to understand both the geophysical setting of the region and past history. The region has been defined by both hydrologic and floristic parameters.

Two common definitions include (1) the area of the western US that is internally drained, with no outlets to the ocean (hydrologic definition) and (2) a floristically defined region dominated by shrub/steppe and woodland plant communities (Pellant et al. 2004). The hydrologic Great Basin covers in excess of 293,000 km^2 and includes much of Nevada and Utah, major portions of Oregon and California, and small areas of Idaho (United States Geological Survey 2013). The floristically defined Great Basin includes more area, with shrub/steppe communities dominated by species of *Artemisia* and *Atriplex*, and woodlands dominated or codominated by species of *Juniperus*. Using either definition, the Great Basin is bounded on the west by the Sierra Nevada and Cascade ranges and on the east by the Rocky Mountains. We will refer to the region as the Great Basin, but will focus our restoration discussion on the western sagebrush steppe or sagebrush semidesert. The sagebrush steppe biome stretches to the Great Plains of the central US, but the shift in precipitation (more summer and less spring and winter) in the eastern portion of the biome makes it ecologically distinct from the western sagebrush steppe.

To provide a sense of restoration challenges, we will break past history of the Great Basin into two segments: (1) recent geologic history (from the end of the Pleistocene about 11,700 years before present) and (2) the period of active Euro-American settlement from about 1850 to present. The geologic history of the Great Basin is important for understanding the physical setting and variability of this region. Toward the end of the Pleistocene, the climate was much cooler and wetter than present day, and there were extensive marshes and lakes in the region. For example, present day Great Salt Lake has a surface area of 4400 km^2 and maximum depth of about 7–10 m depending on the year. The Great Salt Lake is a remnant of the Pleistocene-era Lake Bonneville, which at its peak was almost 52,000 km^2 in size and more than 300 m deep. There were other large Pleistocene lakes, such as Lake Lahontan to the west, which at its peak was equal to 8 % of surface area of the state of Nevada (Nevada Division of Water Resources 2000). As climate dried, these lakes receded, and there was sorting of soil particle sizes along the shorelines. In fact, peak shoreline levels are still visible in portions of the Great Basin. The combination of dramatic climate shifts, internal drainage, and prior geologic activity created an extremely variable environment. Volcanic activity deposited ash layers in portions of the Great Basin and plate tectonic activity resulted in crustal thinning, generating a series of north/south oriented mountain ranges, creating large topographic variation (Fiero 1986). An example of this variability can be seen in a soil map of the Northern Great Basin Experimental Range (NGBER) in southeastern Oregon (Fig. 1, Lentz and Simonson 1986). Although the area is only about 6500 ha, there are 54 soil map units within the experimental range. This high variability makes it difficult to generalize restoration plans, and research must be viewed based on site characteristics associated with a specific research effort.

The period of Euro-American settlement of the region largely began with the gold mining boom in California in the late 1840s and early 1850s (Table 1). This was a period of unprecedented westward migration and created the conditions which led to extensive settlement. By mid-1869, a railroad was completed across the northern Great Basin, allowing transport of people and materials into and out of the region to either the east or west. Because of the arid nature of the region, programs to transfer land from the federal government to private ownership (Homestead Acts) did not function as intended (Svejcar 2015). These programs (initiated in the early 1860s) were developed for the eastern US and not modified adequately for the Great Basin. Thus, much of the land remained in public ownership, and there was no planning for how the lands would be managed. The livestock boom of the late 1880s and lack of oversight on land-use resulted in huge numbers of livestock and serious land degradation (Young and Sparks 1985). A significant drought and harsh winters during this period magnified the overgrazing issue. It was not until the mid-1930s that laws were passed to bring order and management to the publicly owned lands of the Great Basin. The damage inflicted on this arid region resulted in significant restoration efforts, which will be described in a subsequent section.

Two other events that would impact the need for restoration were (1) the introduction of cheatgrass (*Bromus tectorum* L.) and other exotic annuals in the later 1800s and (2) the severe drought of the 1930s. Cheatgrass is an invasive annual grass that is now almost ubiquitous on low and mid-elevation Great Basin rangelands (e.g., Kitchen 2014). This species

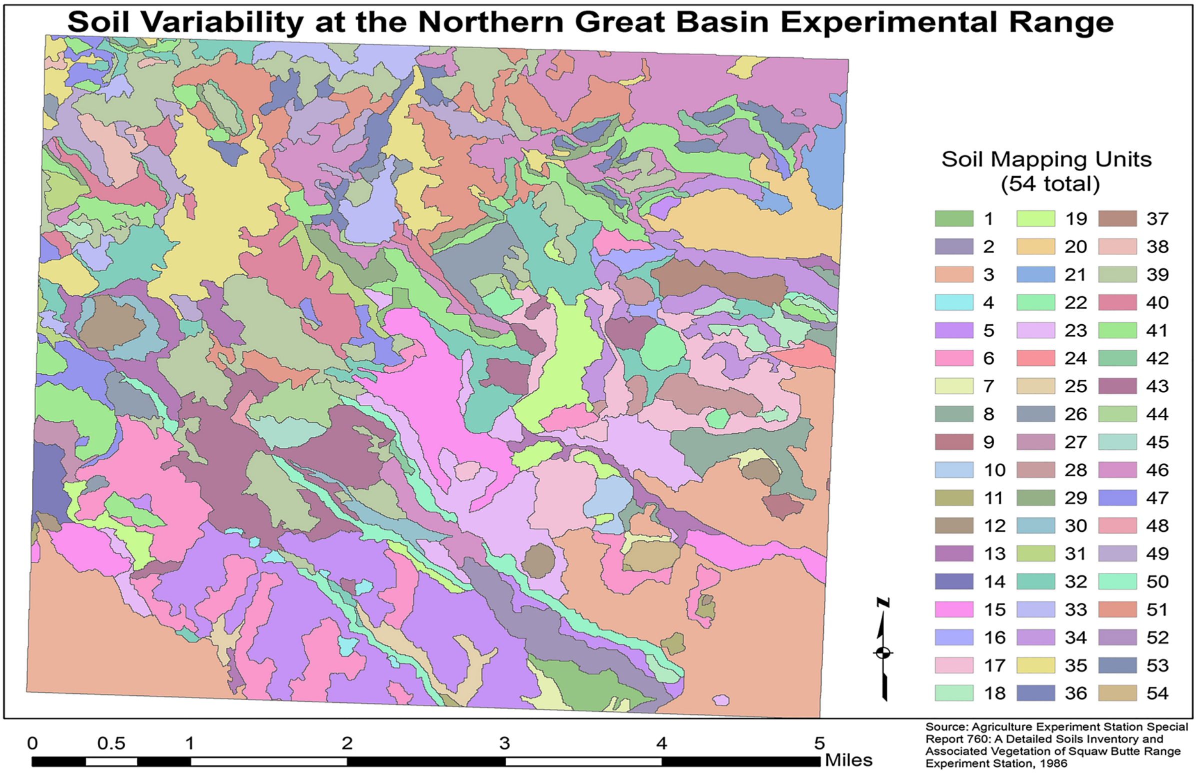

Fig. 1 Soil map of the 6500 ha Northern Great Basin Experimental Range (NGBER)

competes with native seedlings and dramatically increases the number of wildfires in many areas (Whisenant 1990). The drought of the 1930s caused widespread abandonment of homesteads (lands recently transferred from public to private ownership) and put additional stress on native plant communities.

Climate—present and future

The climate of the Great Basin is influenced by its landforms. The Sierra Nevada and Cascade Mountain ranges to the west exert a strong rain shadow effect on much of the region. The large elevational and topographic variation from the north/south mountain ranges (Fiero 1986) also influences climate. The basins generally average less than 25 cm, whereas the higher elevation sites can average over 50 cm in annual precipitation. Bailey (1995) describes the climate of the Intermountain Semidesert and Desert Province (central and southern Great Basin) as being characterized by hot summers and moderately cold winters, with average annual temperatures ranging from 4 to 13 °C. He depicts annual precipitation as ranging from 13 to 49 cm, and often falling as winter snow, with almost no summer precipitation except in the mountains.

Spatial variation in climate is only one part of the challenge faced by vegetation managers and restoration practitioners in the Great Basin. A second major challenge is high annual weather variability (e.g., Fig. 2). Figure 2 represents crop year precipitation at the NGBER over a 70-year period. About one of every 4 years falls within ±10 % of the long-term average and values regularly range from 15 to 45 cm. West (1999) estimated the coefficient of variation in total annual precipitation to be about 30 % for sagebrush steppe ecosystems. The combination of high spatial and temporal variability creates significant challenges for reseeding and other restoration efforts in the region. Projections suggest climate variability will increase in the future (Mote et al. 2013). Some of the projected changes may strongly interact with both size and frequency of wildfires.

Historically, lightening- and human-caused fire was a natural part of western sagebrush steppe ecology

Table 1 General chronology of events, human impacts, restoration focus, and disturbance regimes in the Great Basin, USA

Year	Major events	Phase of human impact	Restoration focus	Disturbance regime
Pre-European settlement		Native American land management utilizing fire and use of native resources through hunting and harvesting		Periodic human and non-human caused fire
1800		Exploration		
1850	California gold rush	Emigration Euro-American settlement		
	Cattle boom and bust	Development of natural resource industries (logging, mining, grazing, agriculture)		Mineral/forage exploitation and conversion of rangeland to farmland
1900	Cheatgrass introduction		Maximize productivity: very input intensive and mechanized; native species not a focus	Decreasing fuels and fire frequency with grazing at high elevations
	Severe drought of 1930s	Homestead abandonment		
1950	Post World War II development of diesel equipment and focus on productivity		Increasing focus on native species, but seeding and plant propagation techniques still agronomic	Mechanized fire suppression post-World War II decreases high elevation fire and increased fire presence in low elevations where annual grasses provide fine fuel continuity
		Urbanization		
2000	Increasing focus on native biodiversity		Increasing focus on natural processes, factors limiting success, and combinations of species	
Present	Increasing CO_2 levels, fire frequency and fire intensity.			Increased size of wildland fires in association with continued spread of annual grasses, increased woody plant fuels, and climate change
Future	Continued alterations in climate			

(Stewart 2002; McAdoo et al. 2013; Kitchen 2016). However, recent climate trends have resulted in a marked increase in the frequency and areal extent of fire across the western US (Westerling et al. 2006; NOAA 2012; National Interagency Fire Center 2013). Modeling efforts have suggested a further expansion and acceleration of fire regimes as warming temperatures and shifts in seasonal precipitation unfold within ongoing climate change (Fule 2008; Yue et al. 2013). Climate, especially precipitation, and associated ecological dynamics across western North America follow annual and decadal variation in the strength of global circulation processes such as the Pacific Decadal Oscillation (PDO) and Northern Annular Mode (NAM) (Hessburg et al. 2005; McAfee and Russell 2008). In much of the area encompassed by sagebrush steppe, overall warming is expected to be accompanied by increasing proportions of cool-season rainfall at the expense of snowpack and an increase in more highly variable summer rainfall (Mote and Salathe 2010; Mote et al. 2013). This will likely result in sagebrush steppe vegetation adapting to a more pronounced "pulsed" ecohydrological regime, altering the spatial and temporal variation in community and ecosystem functioning, and increasing the probability of conditions conducive to fire (Weltzin et al. 2003; Huxman et al. 2004; Rocca et al. 2014).

The effects of climate change in sagebrush steppe ecosystems will be modulated by the ongoing ecological changes associated with shifts in community composition and land-use management. The spread of exotic annual grasses and their acceleration of fire cycles have a well-known degrading effect on sagebrush steppe ecosystems, especially at more xeric,

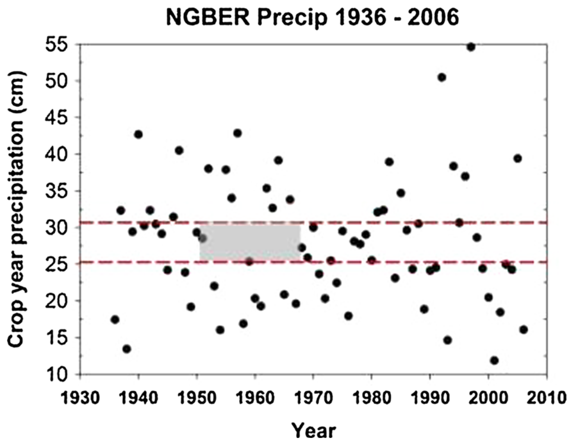

Fig. 2 Crop year precipitation at the Northern Great Basin Experimental Range (NGBER) west of Burns, OR. *Red lines* are ±10 % of the mean. (Color figure online)

lower elevation locations (D'Antonio and Vitousek 1992; Bradley et al. 2006; Davies et al. 2011). Extensive and intensive livestock grazing, as well as extensive fire-suppression following European settlement dramatically altered species abundances and distributions, facilitating the spread of invasive cool-season annual grasses and reducing palatable bunchgrass species; however, implementation of certain management practices, especially changing the seasonal timing and grazing intensity led to a rebound of cool-season bunchgrasses (Miller et al. 1994; Miller and Rose 1999; West 1999). Grass biomass provides critical fuels for spreading fire in sagebrush steppe; however, unlike the case with exotic annual grasses, the role of greater bunchgrass biomass in the recent increases of fire frequency and extent are not well known. Release from overgrazing may have resulted in a more homogenous perennial fuel load across wide areas of sagebrush steppe, a characteristic thought to be critical in facilitating "mega-fires" in response to changing climate in other systems (Bowman et al. 2009).

While changing climatic conditions can produce strong year to year variation, atmospheric carbon dioxide concentrations will continue to rise steadily. Elevated CO_2 will likely increase plant biomass and fuel loads, especially in cheatgrass, which responds more strongly to CO_2 enrichment than do native perennial grasses and forbs (Smith et al. 1987, 2000; Huxman and Smith 2001; Ziska et al. 2005). Biomass of sagebrush seedlings have shown both positive (Johnson and Lincoln 1990) and neutral (Lucash et al. 2005) responses to elevated CO_2. The CO_2 response of adult sagebrush plants or seedlings growing in field settings is currently unknown. Lessons learned from other North American aridland systems likely apply to sagebrush steppe. CO_2 enrichment has its strongest effects on productivity and recruitment in wet years (Hamerlynck et al. 2002; Housman et al. 2003; Naumburg et al. 2003); the gains of which are diminished over prolonged dry periods (Newingham et al. 2013). This suggests that (1) the relative amount of fine fuels and coarse fuels will covary considerably with variation in precipitation and depth and persistence of soil moisture and (2) years immediately following wetter years are likely to have greater fuel loads due to enhanced prior year productivity of both grasses and shrubs. The combination of fire promoting invasive annual grasses and elevated atmospheric CO_2 is a major concern for vegetation managers in the Great Basin because of the potential for increased wildfire frequency. In addition, enhanced wildfire intensity with higher productivity under elevated atmospheric CO_2 could exacerbate the negative effects of altered temperature and precipitation regimes on woody plant recruitment and establishment (Enright et al. 2015). Loss of shrub cover is a significant issue for the maintenance of sagebrush-obligate animal species such as greater sage-grouse.

History of restoration

The initial emphasis on productivity and finding ways for early settlers to survive in the arid Great Basin was largely a failure. Settlers from the eastern and central US had little or no experience with arid lands, and expectations were dramatically inflated by both land speculators and a federal government intent on drawing new settlers to the region. There were many attempts to promote a variety of crop and pasture species and as such, the earliest attempts to seed vegetation in Great Basin plant communities were consistently unsuccessful. Grass seeding trials in the 1890s and early 1900s in the western United States generally failed because seed was only available for cultivated forage plants better adapted to more humid climates (Stoddart et al. 1975). Grasses more adapted to arid and semiarid conditions were needed to successfully establish in these rangelands. Crested wheatgrass (*Agropyron cristatum* [L] Gaertn. and *Agropyron desertorum* [Fisch.] Schult.), an introduced bunchgrass, eventually filled this need in the Great Basin.

Although crested wheatgrass was introduced to North America from Russia in the late 1800s (Young and Clements 2009), seeding this species did not become common until after the 1930s following accelerated erosion of topsoil as a result of drought and farm abandonment (Sharp 1986). Large seeding projects were also impractical until the late 1940s and early 1950s when a durable rangeland plow and seeding drill capable of handling rocks and shrubs were developed (Young and McKenzie 1982); indeed, some of the equipment from this era is still in use today (Fig. 3). Crested wheatgrass was seeded extensively in the Great Basin to compete with halogeton (*Halogeton glomeratus* [Bieb.] C.A. Mey.), an exotic annual forb that is poisonous to sheep, and to increase livestock forage (Miller 1943, 1956; Frischknecht and Harris 1968; Vale 1974). The Halogeton Control Bill of 1952 provided funding to government agencies to seed crested wheatgrass across large expanses of the Great Basin (Young 1988). Crested wheatgrass was often selected over native species because it was less expensive, more available, and established better in drier rangelands than did native bunchgrasses (Robertson et al. 1966; Hull 1974). Many of the areas that were seeded had been overgrazed by livestock resulting in a depleted native herbaceous understory and increases in sagebrush dominance (Vale 1974; Young 1988). Prior to seeding, sagebrush was often removed using fire, mechanical, or herbicide treatments (Vale 1974).

Initially, it was theorized that crested wheatgrass could serve as a bridge species that would occupy a depleted site to prevent further degradation and limit exotic annuals but allow transition to a native-dominated plant community (Cox and Anderson 2004). However, efforts to increase the abundance of native vegetation in crested wheatgrass stands have largely failed because crested wheatgrass rapidly recovers from control treatments (Hulet et al. 2010; Fansler and Mangold 2011) and has more aggressive recruitment than native bunchgrasses (Nafus et al. 2015). Multiple year control may be needed to open crested wheatgrass stands to recruitment of native species, but exotic annuals may take advantage of any decrease in crested wheatgrass cover and density (Hulet et al. 2010). Although concerns with seeding crested wheatgrass have arisen, this species is still frequently seeded after wildfires because of its ability to suppress exotic annual grasses (Arredondo et al. 1998; Davies 2010), relative low cost, and ease of establishment compared to native species (Pellant and Lysne 2005; Boyd and Davies 2010; James et al. 2012; Davies et al. 2015).

Fig. 3 Rangeland drill used in seeding projects in the Great Basin

Opposition to the effects of removing sagebrush and creating near-monocultures of crested wheatgrass resulted in a shift toward preference for more diverse and native plant communities during the latter half of the 1900s (Vale 1974; Pellant and Lysne 2005). An increasing focus on native biodiversity by the general public has fueled research on native plant establishment (Richards et al. 1998). However, seedlings of native vegetation have often failed or only been marginally successful in the Great Basin. Seeding native perennial grasses after wildfires across the Great Basin had little effect on long-term grass cover (Knutson et al. 2014). Similarly, seeding native shrubs after fire did not increase shrub cover or abundance compared with unseeded areas (Lysne and Pellant 2004; Knutson et al. 2014). Seeding of native vegetation has been successful at times, particularly at cooler, higher elevations, and areas receiving greater precipitation (Thompson et al. 2006; Davies et al. 2014). However, at hotter, drier, lower elevation, seeding native vegetation has often failed (e.g., Lysne and Pellant 2004; Boyd and Davies 2010; James and Svejcar 2010; Kyser et al. 2013; Davies and Bates 2014; Davies et al. 2015). Aspect also plays an important role in the likelihood of restoration success (Davies and Bates 2016). South aspects are exceedingly difficult to restore because they are hotter and drier than north aspects, leading to water stress for plants (Van de Water et al. 2002), and are a more favorable environment for exotic annual grass invasion in the Great Basin (Leffler et al. 2013). Invasion by exotic annual species creates additional challenges for successful restoration because these species deplete soil moisture earlier than native vegetation and suppress native species growth (Melgoza et al. 1990). Exotic annuals also develop an annual grass-fire cycle that burns too frequently for native perennial vegetation to persist (D'Antonio and Vitousek 1992; Davies and Svejcar 2008; Davies and Nafus 2013).

Part of the issue with poor establishment of native plants is that rangeland seedings in the Great Basin and the rest of the western United States are largely based on standard agronomic practices that may not address the primary sources of mortality in seeded native vegetation (James et al. 2011). The use of row crop technologies designed for fairly uniform establishment every year may not facilitate native vegetation establishment in an ecosystem noted for extreme temporal and spatial variability in macro- and microenvironmental conditions (Boyd and James 2013; Madsen et al. 2013a; Svejcar 2015). Early practices focused on removing residual vegetation that could compete with seeded species to create a favorable environment for seedling establishment and growth (Vallentine 1977). This included plowing or other mechanical and burning or herbicide treatments to remove native shrubs (Cook 1966; Vallentine 1977). These treatments may counter efforts to restore native plant communities because they may fundamentally alter site characteristics and produce legacy effects (Nafus et al. 2016). Morris et al. (2011) found that native plant communities can require decades to centuries to recover from cultivation. Furthermore, these treatments remove remaining native vegetation, creating a greater restoration deficit. These agronomic-based practices are also restricted to areas that are relatively flat and devoid of trees and significant rock cover. If terrain is too rough for drill seeding, aerial broadcast seeding has been used, but these seedings generally fail in the sagebrush communities of the Great Basin, particularly without additional treatments to improve seed-soil contact (Monsen and Stevens 2004).

Historical seeding of native vegetation has been expensive with very limited success in the hot, dry, lower elevation plant communities of the Great Basin. Success is even less likely on south aspects because of their lower resilience to disturbance and resistance to exotic annual grass invasion (Miller et al. 2014a, 2015). Seeding success in these hot dry communities is very unlikely when site factors dictate that aerial (broadcast) seeding is the only option. Historical practices have not been and will not be adequate to restore many Great Basin plant communities.

Limitations to restoration success

Mechanical and herbicide treatments for removing or reducing nondesired plant species have advanced considerably in recent years (Monaco et al. 2005; Davies 2010; Baruch-Mordo et al. 2013; Miller et al. 2014b; Roundy et al. 2014). However, reduction of nondesired species is only the first step in the restoration process. Establishing or increasing the abundance of desired plant species has experienced comparatively less success and represents a preeminent challenge for restoration practitioners in the Great Basin.

Establishing native bunchgrasses plays a critical role in maintaining site stability (e.g., Pierson et al. 2007) and in reducing annual grass dominance (Davies 2008). These two factors are critical in the eventual development of a native plant community. Thus, focus on the limitations to seedling establishment of native bunchgrasses is a critical step in the landscape restoration process. These native bunchgrasses may not be as long-lived as previously thought (Svejcar et al. 2014), and thus, natural recruitment is also necessary to maintain native plant communities.

Restoration of Great Basin rangeland is set within an environment of extreme variability in space and time (Chambers et al. 2014; Svejcar 2015). Spatial variability is associated with soil factors and complex topography, including the effects of aspect and elevation (Miller et al. 2013). Generally speaking, soil moisture increases and soil temperature decreases with increasing elevation (West and Young 2000). This variation is in turn associated with a positive correlation between elevation and plant production potential (Alexander et al. 1993). The plant production gradient is ecologically significant because resistance of plant communities to annual grass invasion and resilience after fire and other disturbances decreases with decreasing plant production (Chambers et al. 2014). Thus, the need for restoration as well as the degree of threat associated with exotic annual grasses is higher at low versus high elevation sites. Similarly, topographic position interacts with elevation to decrease plant community resilience and resistance to annual grass invasion on warmer and drier aspects (Chambers et al. 2007; Condon et al. 2011). The net effect of this variability is to create a spatially challenging restoration environment in which managers must consider the effects of spatial environmental variation in deciding what techniques will be employed, what plant materials will be used, and how restoration effort will be implemented across the landscape. At large scales, soil moisture and temperature mapping can provide an index of the potential for restoration success or conversion to annual grasses. At more local scales, state and transition models are useful for developing restoration priorities and selecting restoration practices (e.g., Boyd et al. 2014).

The climatic and environmental conditions of the Great Basin create a plethora of challenges for restoration practitioners. Perhaps because of climatic uncertainty, native plants have evolved to invest resources in below-ground biomass at the expense of reproductive biomass, reducing performance of seeds in reaching critical demographic milestones when compared with nonnative cohorts (Madsen et al. 2012a). Thus, there is a stark contrast between the objective of restoration practitioners, to have seeding success at a fixed point in space and time, and the episodically favorable conditions for sexual reproduction to which native Great Basin plants have evolved (Boyd and James 2013). To establish, the seed must be successful through a series of life stage transitions (Fig. 4). With each transition comes a series of factors that can limit successful establishment.

Recent research has shown that the timing of seedling development can also interact with inter-year climate factors to decrease seeding success. For example, seeds of perennial bunchgrass species are typically sown during the fall. Conventional wisdom has been to plant seeds as late as possible during the fall to preclude germination prior to periodic frozen soil conditions during the winter period. However, significant portions (approaching 70 %) of a fall-planted seed population may germinate prior to winter, and seedling emergence, not germination, appears to be the most limiting demographic stage for native perennial bunchgrasses (James et al. 2011; Boyd and James 2013). Subsequent work has demonstrated that germinated but nonemergent seedlings may incur high mortality during frozen soil conditions experienced in winter (Boyd and Lemos 2015). Within-year issues of seedling performance may be partially overcome by adjusting timing of planting. For example, Boyd and James (2013) found that in years with adequate rainfall, early fall planting (September–October) yielded highest spring seedling densities. Spring planting may allow seedlings to develop after periods of frozen soil during winter (Boyd and Lemos 2015). However, spring planting conditions (wet soils following snow melt) often preclude planting with currently available ground-based machinery.

Seeding success may also be limited by planting method. Currently, most seeding in the Great Basin region utilizes drill (Fig. 3) or broadcast seeding. Broadcast seeding in mesic mountain big sagebrush plant communities can be successful for both shrub and grass species (Davies et al. 2014). However, broadcast seeding in lower elevation annual grass-prone sites has had only limited success (Lysne and Pellant 2004). Drill seeding offers improved seed-soil

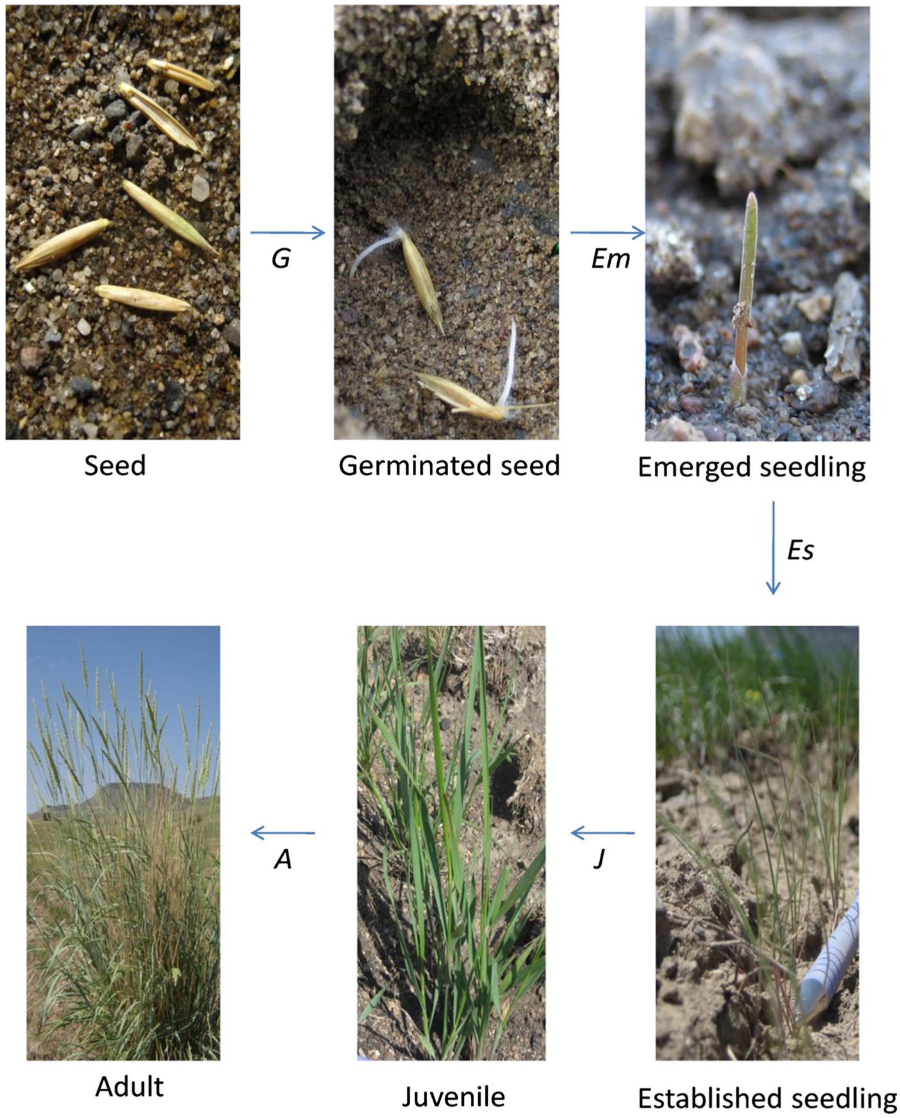

Fig. 4 Pictoral representation of the life stages of a rangeland bunchgrass. *G* germination, *Em* seedling emergence, *Es* seedling establishment, *J* juvenile, and *A* adult stage (Svejcar et al. 2014)

contact relative to broadcast methods, but the results have been mixed at best, and determining the efficacy of past efforts is clouded by an apparent literature bias toward publication of results from seedings undertaken in years of above average precipitation (Hardegree et al. 2011). Working in Wyoming big sagebrush plant communities, James and Svejcar (2010) found that hand-seeding to exact depth in the fall following summer wildfire increased resulting seedling density over 7-fold relative to drill seeding, and thus seeding technology was a much greater barrier to seedling establishment than competition from exotic weeds.

Successful seeding in the Great Basin is contingent on overcoming both environmental and planting technique limitations. Recent advances in seed enhancement technology show promise for helping managers to navigate such barriers. For example, seed coatings have the potential to delay germination of fall-planted bluebunch wheatgrass (*Pseudoroegneria spicata* (Pursh) A. Löve) seed until spring and more than double resultant spring seedling density (Madsen et al. 2016). Increases in seedling density have also been demonstrated with surfactant-coated seeds, which allow for root penetration of hydrophobic soil layers that often develop after wildfires in woodlands (Madsen et al. 2013b). In addition to factors associated with soil moisture and temperature, other potentially limiting factors such as soil crusting may be ameliorated through the use of seed enhancement technologies. Madsen et al. (2012b) found that agglomeration of multiple seeds into seed pellets increased perennial grass seedling emergence through high-clay soils; this technology may be particularly beneficial to small-seeded species such as Wyoming big sagebrush (Madsen et al. 2016). Genetic selection for specific plant traits may also be a tool for improving native plant establishment (Leger and Baughman 2015).

Sociopolitical challenges

Seeding in the Great Basin may also be constrained by interaction between the complex nature of the seeding environment and regional/national policies that focus on implementation of practices. Boyd and Svejcar (2009) defined types of problems in natural resources management based on degree of complexity. “Simple” problems were defined as those problems that have a limited number of causal factors and for which the nature of the solution did not vary appreciably over space and time (e.g., discharge of effluent into water bodies). “Complex” problems, in turn, have multiple and often interacting causal factors such that the nature of the “solution” varies depending on factors that are dynamic in space and time (e.g., predicting plant productivity). While restoration in the Great Basin is a complex problem, regional and national programs that support restoration activities are often created around specific practices, the “success” of which is tallied based on money spent and hectares treated within a program. Such tendencies run counter to the dynamic nature of complex problems, and addressing these discrepancies will involve finding ways to increase flexibility in the implementation of restoration activities at local scales to allow for adaptive management, the success of which should ultimately be evaluated based on biological (e.g., seedling or mature plant density) versus programmatic metrics.

We maintain that successful arid land restoration will require a multitiered approach. The first tier is to identify the factors limiting successful seedling establishment. By definition, arid lands are water limited, but clearly other factors come into play. Species that propagate via sexual reproduction would not exist in a community without successful recruitment. Identifying the conditions under which natural recruitment occurs may be a first step in identifying barriers to restoration (Hardegree et al. 2012; Svejcar et al. 2014). The second tier is developing methods to overcome the variable environment. The solution may involve introducing artificial dormancy so that autumn seeded species will not all germinate under favorable conditions and freeze during the winter (Boyd and Lemos 2015), or conversely speeding up germination to allow seedlings to achieve sufficient size to survive the winter. Each region and species group will experience different obstacles (e.g., Madsen et al. 2016), but recognizing the fundamental ecological principals that underlie restoration success locally can facilitate a proactive, adaptive management approach that can be applied regionally (Boyd and Svejcar 2009). Accelerating research and scaling up the application of these practices is critical because at current levels of degradation, it is becoming increasingly difficult to manage losses of native plant communities and the habitat that they provide.

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