Rodent herbivory and fire differentially affect plant species recruitment based on variability in life history traits

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Abstract. Rodent consumers can have strong top-down effects on plant community development via seed predation, but their influence on seedling recruitment and how it varies depending on disturbance history and plant traits are largely unknown. Over a two-year period, we experimentally tested the influence of rodents and fire on seedling survival of 14 plant species with contrasting growth forms and life history traits in the Great Basin Desert. Seedling survival was dramatically increased by rodent exclusion and fire and was greater in 2013 than 2012. Rodent abundance was an order of magnitude lower in 2013 than 2012 and across both years showed a negative linear relationship with mean seedling survival of all 14 plant species ($R^2 = -0.62$). There was strong variability in plant species sensitivity to rodent herbivory with a more than 10-fold difference in mortality risk between the most sensitive species and the most resistant species and a high degree of variability in survival among all species. Mortality risk of forb species was double that of grass species and there was a trend toward higher mortality of native species compared to introduced species. The results suggest that rodent herbivory may be an important determinant of plant community assembly in desert ecosystems with plant life history strategies and growth form traits serving as important survival filters. Post-fire environments and low rodent years likely provide windows of opportunity for greater seedling recruitment due to a release from rodent herbivory pressure.

Key words: desert; fire; folivory; Great Basin; herbivory; seedling; small mammals; top-down.

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INTRODUCTION

Plant communities are rapidly changing across earth’s biomes due to land use change, plant invasions, and expansion of wildfire (Bowman et al. 2009, 2011). Successful post-disturbance recruitment of native vegetation has been identified as a primary hurdle to re-establishing ecosystem stability and function in response to these human-altered disturbance regimes (Pearson et al. 2016). While it is well established that abiotic limitations are critical constraints to recruitment of desirable plant species following disturbance (Abbott and Roundy 2003), increasing evidence suggests that biotic factors also play a critical role. Among the most important biotic filters on plant community development following disturbance is herbivory by animal consumers (Zwolak et al. 2010, Horn et al. 2012, Wan et al. 2014).

Rodents function as ecosystem engineers across earth’s biomes (Reichman and Seabloom 2002, Zhang et al. 2003). Rodents can have strong top-down effects on the establishment success of native and invasive plant species (Brown and Heske 1990a, Maron and Crone 2006). We recently documented that experimental exclusion of rodents can cause drastic shifts in plant community composition, diversity, and invasion outcomes in deserts (St. Clair et al. 2016). Rodents influence plant community assembly by two potential mechanisms: consuming seeds (granivory) or through the consumption of seedlings or...
vegetative tissues (folivory). There is a large literature detailing how differential seed preference by rodents based on seed quality and seed size alters patterns of plant community development (Maron and Crone 2006, Maron et al. 2012, Connolly et al. 2014). We know far less about the role that rodents have on plant community assembly through folivory. Rodents have been shown to reduce plant growth and survival in single-species experiments (Pyke 1986, Duval et al. 2005, Meyer and Pendleton 2005, Bestelmeyer et al. 2007), but we lack multi-species studies under natural conditions that test how variability in plant traits and disturbance history impacts the influence that rodents have on seedling survival and plant community assembly.

Rodent populations fluctuate across space and time (Brown and Heske 1990b, St. Clair et al. 2016) including periods of low rodent abundance that may be critical periods for plant establishment. In contrast, periods of high rodent activity may severely limit seedling recruitment (Shea and Chesson 2002, Allington et al. 2013, St. Clair et al. 2016). Disturbance may contribute to these spatio-temporal fluctuations in the abundance and diversity of rodent communities (Beatley 1976, Simons 1991, Horn et al. 2012, Freeman et al. 2014). In arid ecosystems, bipedal rodents (i.e., *Dipodomys* spp.) typically prefer foraging in inter-shrub spaces (Rosenzweig and Winakur 1969, Kotler 1984) and therefore tend to have positive responses to reduction in plant cover following disturbance (Simons 1991, Monasmith et al. 2010, Horn et al. 2012). In contrast, quadrupedal species (e.g., *Peromyscus* spp., *Perognathus* spp.) tend to forage under and near shrubs (Rosenzweig and Winakur 1969, Kotler 1984, Falkenberg and Clarke 1998) as evidenced by their reduced abundance with losses of plant cover in post-fire landscapes (Groves and Steenhof 1988, Simons 1991, Ostoja and Schupp 2009, Horn et al. 2012). Fire may therefore indirectly mediate changes in plant community development by altering the composition and abundance of rodent communities and their top-down effects on seeds and seedling fate.

The re-establishment of vegetation in post-fire environments has rarely been considered in the context of herbivory by consumers particularly rodents that can have strong top-down controls on the development of plant communities, especially in arid ecosystems. The objective of this study was to evaluate the influence of rodent consumers and fire on the seedling survival of desert plant species in a semi-arid desert. The following hypotheses were tested: (1) Rodent herbivory reduces plant establishment, an effect that varies depending on rodent abundance; (2) post-fire conditions decrease herbivory risk of seedlings by modifying the rodent community; (3) plant establishment patterns are influenced by differential species susceptibility to rodent herbivory due to variation in plant traits.

**MATERIALS AND METHODS**

This study was conducted in the Great Basin Desert in Rush Valley, Utah, USA (40°05′26.17″ N 112°18′18.01″ W, elevation: 1650 m). Long-term mean average temperature for Rush Valley is 8.6°C with strong seasonal variation. Average temperature for the month of January is −3.2°C, and average temperature for July is 22.3°C (Vernon GHCN: COOP, Utah Climate Center). Average annual precipitation is 257 mm/yr with a relatively even distribution throughout the year. The soil at our study location is classified as a silty, mixed mesic Haplic Natrargid, Taylors Flat Loam. The study site had little evidence of grazing and no evidence of wildfire based on intact soil crusts and a well-developed perennial shrub community. *Artemisia tridentata* subsp. *wyomingensis* (Wyoming sagebrush) and *Elymus elymoides*, a native perennial bunch grass, were the dominant plant species at the study site. The most common rodent species were *Peromyscus maniculatus*, *Dipodomys microps*, and *Perognathus parvus*.

**Experimental design**

The study was designed to test the main effects and interaction of burned habitat and rodent exclusion on the seedling fate of 14 desert plant species with contrasting growth form and life history traits. The study was a full factorial design in five replicated blocks. We randomly assigned all four treatment combinations (+rodents +fire, −rodents +fire, +rodents −fire, −rodents −fire) within each of five experimental blocks. The five experimental blocks were evenly spread over a 7-ha area with a wire fence established around the perimeter to exclude livestock but with space at the bottom of the fence to allow free movement of native mammals and reptiles. Each 60 × 60 m block was
divided into four quadrants (30 × 30 m). Blocks were 30 m apart from one another. Rodent fences were installed using 1.0-m-tall welded wire fencing in a 60 × 60 m grid with four 30 × 30 m quadrants in June 2011. The fencing was buried 30 cm below the soil surface and it extended 70 cm aboveground. Rodent exclusion plots had 20 cm of flashing installed on the top of the fence to inhibit climbing by rodents. Rodent control plots had 12 × 10 cm openings every 4 m at the base of the wire fencing to permit free movement of rodents in and out of the plots.

The burn treatments were conducted on 20 September 2011. To facilitate the spread of fire between the large shrub interspaces, we placed 300 g/m² of wheat straw between shrubs according to the methods of Esque et al. (2010). The fires were initiated with drip torches, and burn severity was high with >99 percent plant mortality (St. Clair et al. 2016). This is typical of the burn characteristics of mature sagebrush communities in late summer. Plant cover in unburned plots in 2012 and 2013 was approximately 35% and was predominately sagebrush. Plant cover in burned plots was <2% in 2012 and <10% in 2013 and was dominated by two invasive annual plant species Bromus tectorum (cheatgrass) and Halogeton glomeratus (St. Clair et al. 2016).

Preliminary trials in the greenhouse were conducted to determine the length of time each plant species needed to grow to a mean height of 2.5 cm. Results from this trial were used to determine planting times and germination rates that resulted in the establishment of 15 plants per pot timed to match the emergence of seedlings on the landscape at our study site. A mix of species was chosen that included native and introduced species often used for stabilization and rehabilitation treatments by the Salt Lake City Bureau of Land Management office. Each species was grown from seed in a 10 × 10 cm plastic pot in a mixture of 50% sand, 25% perlite, and 25% sieved top soil from the study site.

Emerged seedlings in pots were placed into the experimental plots in mid-May of 2012, approximately 8 months after the fire treatment was applied, and again in mid-May 2013. Seedling pots were buried up to the rim so they were even with ground level. Fourteen pots (one per species) with 15 plants each were placed 0.5 m apart in random order within each plot. Seedling pots were placed at least 4 m distance from fence lines to avoid edge effects. The number of plants alive per pot was recorded each day for 7 d from the beginning of the experiment. Plants were counted as dead if no whole leaves remained, though most plants subject to herbivory had no leaves and little to no remaining stem above soil level. While the collection of survival data ended after 7 d, all pots with seedlings remained in the experimental plots for 10 more days. During the initial seven-day assessment period and the 10 d that followed, there was no evidence of seedling resprouting.

Rodent trapping
We surveyed rodents in both years after conducting the seedling survival surveys; this was done in August of 2012 and July of 2013. Eight Sherman live traps were placed inside each experimental plot during the rodent survey period. Each trap was baited with commercially available rodent seed mix (Pennington Seed, Madison, Georgia, USA). Traps were set each evening and checked each morning for three consecutive nights. Captured animals were marked with individually numbered ear tags. All animals were released at the point of capture unless they were trapped inside an exclosure plot in which case they were released at the nearest point outside the exclosure.

Statistical analysis
Plant survival data were divided into two data-sets, one for each year of the study, and each dataset was analyzed separately. Mixed effects Cox proportional hazards analysis was used to test the main and interactive effects of rodent exclusion, fire, and plant species (fixed effects) on seedling survival over the seven-day plant survey period. Block was included as a random term in the models. We selected the best model using Akaike’s information criterion, corrected for sample sizes using the coxme package in the program R version 3.1.1 (Therneau 2012a, R Core Team 2014). The top model from the mixed effects analysis for each dataset was recreated as a fixed effect Cox proportional hazards model using the survival package in R with block included as a fixed factor (Therneau 2012b). Fixed effect models were used to make pairwise comparisons of treatment effects on each plant species using the lsmeans package in R (Lenth 2016). Pairwise comparisons were made among treatments (unburned-excluded, unburned-present, unburned-present, burned-present, burned-excluded) and plant species (fixed factors) and were done in R using the lsmeans package.
burned-excluded, and burned-present) for each plant species individually and for all species combined. A logit-transformed linear mixed model (lme4 package in R) was used to test the main and interactive effects of fire, rodent exclusion, and year on the mean seedling survival of all 14 plant species (Bates et al. 2014). Fire, rodent exclusion, and year were characterized as fixed effects, and block was included as a random effect. Analysis of variance was used to test the effects of year and burn condition on rodent abundance and to perform contrasts of forbs vs. grass species and native vs. introduced species with block included as a random term using function lmer in package lme4 in program R. Simple linear regression was used to examine the relationship between rodent abundance and seedling survival of all 14 species for both years. Analysis of variance and simple linear regression tests were performed using JMP statistical software (SAS, Cary, North Carolina, USA).

**RESULTS**

**Seedling survival**

Seedling survival was significantly affected by rodent exclusion and fire with sensitivity varying strongly between plant species and across years (Tables 1, 2). Mean seedling survival averaged across all 14 plant species was significantly higher in rodent exclusion plots and burned plots compared to control plots and was greater in 2013 than 2012 (Fig. 1; Appendix S1: Table S1). The year-by-fire and year-by-rodent exclusion interaction terms indicate that rodent and fire effects were much stronger in 2012 than 2013 (Fig. 1; Appendix S1: Table S1). In 2012, averaged seedling survival rates of all 14 species was fourfold greater in burned plots compared to unburned control plots (33% vs. 8% in 2012), and fivefold greater in rodent exclusion plots compared to control plots (35% vs. 7% in 2012). Seedling survival

| Species                          | Functional group | Origin       | Unburned Excluded | Unburned Present | Burned Excluded | Burned Present |
|----------------------------------|------------------|--------------|-------------------|------------------|-----------------|----------------|
| *Purshia glandulosa* (desert bitterbrush) | Forb             | Native       | 1.20 ± 0.14^A     | 2.55 ± 0.15^B    | 0.10 ± 0.22^C   | 1.57 ± 0.17^A  |
| *Artemisia tridentata* (sagebrush)   | Forb             | Native       | 1.75 ± 0.12^A     | 2.69 ± 0.12^B    | 0.96 ± 0.13^C   | 1.36 ± 0.12^AC |
| *Medicago sativa* (alfalfa)         | Forb Introduced  |              | 1.28 ± 0.12^A     | 3.02 ± 0.12^B    | 0.62 ± 0.13^C   | 1.34 ± 0.12^A  |
| *Atriplex canescens* (four-wing saltbush) | Forb             | Native       | 1.44 ± 0.13^A     | 3.02 ± 0.12^B    | −0.12 ± 0.18^C  | 1.20 ± 0.12^A  |
| *Achillea millefolium* (yarrow)     | Forb             | Native       | 1.36 ± 0.13^A     | 2.26 ± 0.12^B    | 0.13 ± 0.14^C   | 1.00 ± 0.12^A  |
| *Linum lewisii* (Lewis flax)        | Forb             | Native       | 1.67 ± 0.12^A     | 2.50 ± 0.12^B    | −0.08 ± 0.16^C  | 0.93 ± 0.12^A  |
| *Poa secunda* spp. *canbyi* (Canby bluegrass) | Grass           | Native       | 1.37 ± 0.12^A     | 2.20 ± 0.12^B    | 0.48 ± 0.14^C   | 0.88 ± 0.13^C  |
| *Agropyron fragile* (Siberian wheatgrass) | Grass           | Introduced   | 0.79 ± 0.14^A     | 2.11 ± 0.12^B    | −1.19 ± 0.25^C  | 0.83 ± 0.12^A  |
| *Agropyron cristatum* (crested wheatgrass) | Grass           | Introduced   | 0.95 ± 0.13^A     | 2.59 ± 0.12^B    | −1.05 ± 0.24^C  | 0.82 ± 0.13^A  |
| *Pascopyrum smithii* (western wheatgrass) | Grass           | Native       | 0.45 ± 0.14^A     | 2.60 ± 0.12^B    | −0.94 ± 0.22^C  | 0.72 ± 0.14^A  |
| *Elymus trachycaulus* (Snake River wheatgrass) | Grass           | Introduced   | 0.49 ± 0.14^A     | 2.14 ± 0.12^B    | −0.45 ± 0.20^C  | 0.60 ± 0.13^A  |
| *Pseudoroegneria spicata* (bluebunch wheatgrass) | Grass           | Introduced   | 0.21 ± 0.15^A     | 1.55 ± 0.12^B    | −1.57 ± 0.29^C  | 0.55 ± 0.14^A  |
| *Poa secunda* (Sandberg bluegrass)  | Grass           | Native       | 0.89 ± 0.12^AC    | 1.30 ± 0.12^A    | −0.25 ± 0.16^B  | 0.49 ± 0.13^C  |
| *Psathyrostachys juncea* (Russian wildrye) | Grass           | Introduced   | 0.28 ± 0.15^A     | 1.87 ± 0.12^B    | −1.18 ± 0.24^C  | 0.15 ± 0.14^A  |
| All species                      |                  |              | 1.01 ± 0.13^A     | 2.31 ± 0.13^B    | −0.32 ± 0.20^C  | 0.88 ± 0.10^A  |

*Notes: * Means with different letters are statistically different (P < 0.05) within each species. Values are sorted from highest to lowest based on the hazard values in burned plots open to rodents. Higher hazard values correlate with lower survival.

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**Table 1. Hazards (mean ± standard error) from Cox proportional hazards models for seedling survival in 2012 for each treatment.**
increased more than threefold from 2012 to 2013 (20–66%), and while the magnitude of the positive effects of fire and rodent exclusion decreased in 2013, they were still present (fire: 72% vs. 59%; rodent exclusion: 73% vs. 59%).

**Rodent abundance**

There was no significant difference in rodent abundance between burned and unburned plots in either year of the study (Fig. 2). Rodent abundance dropped 11-fold from 2012 to 2013 (Fig. 2) and was negatively correlated with mean seedling survival of all 14 plant species (Fig. 3). Based on the slope of the regression equation, mean weekly seedling survival was reduced 11% with each additional rodent present in the experimental plots (Fig. 3).

**Species sensitivity**

There was strong variability in plant species sensitivity to rodent herbivory (Tables 1, 2; Appendix S1: Table S2), which was most clearly demonstrated in 2012 due to the much greater rodent abundance that year. In our hazards analysis, the most sensitive species (*Purshia glandulosa*) was 10 times more likely to experience mortality in burn plots with rodents present than the least sensitive species (*Psathyrostachus juncea*).

### Table 2. Results of model selection of Cox proportional hazards models.

| Model          | No. parameters | ΔAIC<sub>c</sub> | w<sub>i</sub> |
|----------------|----------------|-----------------|-------------|
| **2012**       |                |                 |             |
| M × B × S      | 56             | 0               | 1           |
| M × S + B      | 29             | 50.28           | 0           |
| **2013**       |                |                 |             |
| M × B × S      | 56             | 0               | 1           |
| M × S + B      | 29             | 220.55          | 0           |

Notes: AIC<sub>c</sub>, Akaike’s information criterion, corrected for sample sizes. Models were compared within each dataset (year). All combinations of the three fixed factors (B = plots with or without fire treatment; M = rodents present or absent; and S = plant species) with a random block factor were modeled; only the top two models for each dataset are shown. Akaike weights (w<sub>i</sub>) give the probability that a given model is the best of those tested to explain patterns in each dataset.

Fig. 1. Mean seedling survival of all 14 plant species per week under factorial treatment combinations of fire and rodent presence in the experimental plots in 2012 and 2013. The main effects of fire, rodents, and year, and the rodents-by-year interaction were the significant factors in the model.

Fig. 2. Mean rodent abundance in rodent access plots as a function of burn condition and year.

Fig. 3. Regression analysis characterizing the relationship between rodent abundance and mean seedling survival of all 14 plant species in the experimental plots over 2012 and 2013.
and there was a high degree of variability in sensitivity among the other 12 species (Table 1). Mortality risk of forb species was double that of grass species and there was weak evidence of higher mortality of native species compared to introduced species (Fig. 4).

**DISCUSSION**

The importance of rodent consumers on the development of plant communities is increasingly being recognized (Maron et al. 2012). Our study provides new insights into the importance of seedling herbivory as a mechanism by which rodents influence plant recruitment success and shows that fire and plant traits can modify these effects. These results generally support our first hypothesis that rodents strongly regulate seedling survival across a range of plant species and functional groups and suggest that fire can have a modifying influence on seedling survival (hypothesis 2). Consistent with our third hypothesis, there was a high degree of variability between species in susceptibility to rodent herbivory, suggesting that seedling preference by rodents related to plant trait differences has the potential to influence patterns of plant community assembly.

**Rodent herbivory**

Rodent consumers can regulate plant community development (Brown and Heske 1990a) and alter plant invasion patterns (St. Clair et al. 2016) through seed predation (Pearson et al. 2011, 2014, Maron et al. 2012). While rodents also utilize vegetative plant tissues in their diet (Hulme 1994, Bestelmeyer et al. 2007), the impacts of seedling herbivory by rodents as a filter on plant community assembly are not as well characterized. This study demonstrates that seedling herbivory by rodents can suppress plant establishment success across a broad group of plant species (Fig. 1, Table 1). Consistent with our results, seedling predation by rodents was a leading cause of mortality of creosote bush a dominant shrub species in deserts of North America (Meyer and Pendleton 2005). Furthermore, the observed differences in plant species sensitivity to rodent herbivory (Table 1) along with granivory appear to be a likely mechanism driving shifts in plant species composition and diversity in our study system (St. Clair et al. 2016).

Rodent populations fluctuate across space and time due to climate variability, disturbance, and reproductive cycles (Brown and Heske 1990b, St. Clair et al. 2016). This was evident by the nearly ten-fold decrease in rodent abundance from 2012 to 2013 (Fig. 3) that was likely the driving force behind the dramatic increase in seedling survival rates from 2012 to 2013 (Fig. 1). This conclusion is further supported by the strong negative correlation observed between rodent abundance and seedling survival (Fig. 3). These results suggest that periods of low rodent abundance in desert landscapes may create windows of opportunity for re-establishment of plant species in post-disturbance landscapes (Allington et al. 2013). However, exotic plant species are also released from consumer-mediated biotic resistance during periods of low rodent abundance, which can
dramatically increase invasion rates leading to competitive exclusion of desirable plant species (St. Clair et al. 2016).

Fire effects

Fire can indirectly affect plant survival by altering the abundance and activity of rodents (Zwolak et al. 2010, Horn et al. 2012, St. Clair et al. 2016). Hazards analysis indicated that when averaged across all 14 species, mortality risk was highest in rodent control plots particularly in the absence of fire (Table 1). This indicates that if seed germination is successful, rodent herbivory can be a significant barrier to seedling establishment, but the effect is attenuated in burned habitats. The positive effects of fire on seedling survival (Fig. 1) would suggest that the rodent community in this study system, which is dominated by quadrupedal species, would be decreased in burned plots due to losses of plant cover following fire. However, our three-day rodent community survey following the seedling survival experiment showed no difference in rodent abundance or species richness in burned and unburned plots (Fig. 2), though rodent diversity decreased in burned plots. A much more comprehensive rodent community survey in spring, summer, and fall from 2011 to 2015 showed a nearly twofold reduction in rodent community abundance, richness, and diversity in these same experimental plots. While burned plots did not appear to have reduced levels of rodent abundance during the period of this study (May), it is possible that disturbance and changes in vegetation structure may have modified the foraging behavior and movement of the rodents as observed in other studies (Skliba et al. 2008, Ostoja and Schupp 2009).

Plant species responses

Plant species sensitivity to rodent herbivory was highly variable in this study (Table 1), suggesting that variation in plant traits created strong differences in rodent preference. Seed selection by rodents is strongly driven by seed size and chemistry (Henderson 1990, Maron et al. 2012, Connolly et al. 2014). The physical and chemical traits of vegetative tissues also influence foraging behavior and species selection by herbivores (Perez-Harguindeguy et al. 2003, Carmona et al. 2011, Lindroth and St. Clair 2013, Villalba et al. 2014). However, most of these studies have looked at herbivory effects on post-seedling stages of plant development and relatively few studies have looked specifically at the effects of rodent consumers. Studies that have documented rodent effects on seedlings have typically been single-species studies (Pyke 1986, Duval et al. 2005, Meyer and Pendleton 2005, Bestelmeyer et al. 2007). Rodent preference between plant species and among forbs and grass species (Fig. 4) observed in this study (Table 1, Fig. 2) is likely influenced by variation in morphological characteristics (Carmona et al. 2011). Hulme (1994) found that seedling size was an important predictor of seedling selection by rodents. Because of nitrogen and water limitation in desert ecosystems, variation in nitrogen and water content in seedling tissue is also likely to influence preference patterns by rodents (Perez-Harguindeguy et al. 2003). Stronger rodent selection of forbs in our study (Fig. 4) is likely influenced by higher forage quality of forbs compared to grasses due to higher tissue nutrient content, better digestibility, and lower levels of silica (Johnston and Bezeau 1962, Gonzalez-Hernandez and Silva-Pando 1999, Cubera et al. 2009). High silica content and lower digestibility of grasses can lead to inefficiency in energy conversion by rodents that translates into slower growth rates and higher levels of mortality (Batzli and Cole 1979, Massey and Hartley 2006, Massey et al. 2007).

Mojave Desert comparison

This study was conducted concurrently with a similar experiment in the Mojave Desert (Sharp Bowman et al. 2017a). In both study systems, rodents greatly reduced seedling survival and changes in rodent abundance through time were associated with observable differences in plant mortality rates. The effect of fire was greater in the Great Basin, perhaps because the rodent community was comprised primarily of quadrupedal rodents, while the Mojave community was comprised of primarily bipedal rodents (Sharp Bowman et al. 2017b). As quadrupedal rodents prefer foraging under shrub cover (Falkenberg and Clarke 1998), their foraging patterns likely are altered more by fire than those of quadrupedal species that prefer open habitat (Rosenzweig and Winakur 1969, Kotler 1984). In both study systems, plant species demonstrated a wide range of
susceptibility to rodent herbivory and this varied among growth forms (grass, forb, shrub). The effects of rodent herbivory and fire history on seedling survival were similar in these two regions and may stretch beyond these study systems.

**Conclusions and Implications**

The direct cost of wildfires in the United States now averages $1.5 billion annually with rehabilitation and indirect costs estimated to be significantly higher (Western Forestry Leadership coalition 2009). Action plans for management of wildfire are most successful and cost-effective when informed by science-driven knowledge of the fire ecology that operates within ecosystems. Data from this study demonstrate that seedling herbivory by rodents is an important constraint to plant recruitment across a broad group of plant taxa. This information is particularly important as invasive grasses continue to drive the expansion of wildfires in North American desert landscapes (Brooks et al. 2004).

Information on rodent community characteristics can help identify windows of opportunity for post-fire re-establishment success of plants. When determining species to be used for post-fire re-establishment of plants, many factors must be taken into account including seed availability, germination rates, pre-fire vegetation, and climatic factors, and, based on our data, sensitivity to rodent herbivory. Just as understanding the impact of rodent granivory can increase germination success (Longland and Ostoja 2013), accounting for rodent herbivory of seedlings and how it varies between plant species is likely to produce more desirable vegetation outcomes. Ecosystem stabilization strategies should also consider the rodent community in the context of disturbance history of the landscape. Rodent community abundance and diversity can efficiently be gathered using short-term trapping surveys. Seeding campaigns following desert wildfires may be successfully used particularly during periods of lower rodent abundance and with a greater proportion of quadrupedal than bipedal individuals (quadrupedal species often decline in abundance after fire; Ostoja and Schupp 2009, Horn et al. 2012). Finally, seeds of species known to be sensitive to rodent herbivory could be augmented in seed mixes to compensate for mortality due to herbivory.

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**Supporting Information**

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.2016/full