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Title: Post-fire ponderosa pine regeneration with and without planting in Arizona and New Mexico

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Abstract:

Forest fires are increasing in size and severity globally, yet the roles of natural and artificial regeneration in promoting forest recovery are poorly understood. Post-fire regeneration of ponderosa pine (*Pinus ponderosa*, Lawson and C. Lawson) in the southwestern U.S. is slow, episodic, and difficult to predict. Planting of ponderosa pine after wildfire may accelerate reforestation, but little is known about survival of plantings and the amount of post-fire natural regeneration. We compared ponderosa pine regeneration between paired planted and unplanted plots at eight sites in Arizona and New Mexico that recently (2002-2005) burned severely. Two sites had no natural regeneration and no survival of planted seedlings. Seedling presence increased with number of years since burning across all plots, was positively associated with forb and litter cover on planted plots, and was positively associated with litter cover on unplanted plots. Survival of planted seedlings, measured five to eight years after planting, averaged 25% (SE = 8) and varied from 0 to 70% across sites resulting in seedling densities of 0 to 521 trees ha⁻¹. Based on a projected 44% survival of seedlings to mature trees and target density of mature trees determined by historical range of variability and ecological restoration principles, four of eight sites have a seedling density in planted plots (125 to 240 ha⁻¹) that will produce a density of mature trees (55 to 106 ha⁻¹) close to desired levels, whereas seedlings are currently deficient at three planted sites, and in surplus at one site, which had abundant natural regeneration. Natural regeneration in unplanted plots during the first decade

after burning produced seedling densities inconsistent with desired numbers of mature trees. Natural regeneration in unplanted plots produced less than 33 seedlings ha⁻¹ at seven of eight sites, but produced 1433 seedlings ha⁻¹ at one high-elevation site that supported a more mesic vegetation community before burning than the other sites. Our results show that current practices for planting ponderosa pine after severe fires in Arizona and New Mexico produce desired numbers of seedlings in approximately half of all projects, whereas natural regeneration rarely does within the first decade after burning.

Keywords: reforestation, wildfire, *Pinus ponderosa*, seedling establishment

1. Introduction

The occurrence of large forest fires has increased globally (Williams et al., 2013) and is projected to continue increasing due to climate warming (Liu et al., 2010; Flannigan et al., 2013; Xu et al., 2013). For example, fire size and perhaps severity have already increased over the last century in forests of western North America (Westerling et al., 2006; Littell et al., 2009; Miller et al., 2009; Westerling et al., 2011). Large intense wildfires have the potential to shift forest to other vegetation types when local tree seed sources are eliminated (Savage and Mast, 2005; Roccaforte et al., 2012). Deforestation caused by wildfire is especially pronounced in semi-arid forests of tree species lacking serotinous cones and resprouting capacity, and where tree establishment is limited by water stress (Pausas et al., 2008; Williams et al., 2010; Savage et al., 2013). Limitations to tree establishment by water and heat stresses likely will be amplified by increasing temperature over the next century (Castro et al., 2005; Herrero et al., 2013; IPCC 2013). Whereas several studies have investigated natural regeneration of trees after wildfire

(Bonnet et al., 2005; Donato et al., 2009; Haire and McGarigal, 2010; Crotteau et al., 2013), few also have considered the role of tree planting in accelerating forest recovery.

Post-fire tree planting is currently controversial. Long delays in tree natural regeneration after intense burning have prompted recommendations for planting native trees (Beschta et al., 2004; Sessions et al., 2004; Passovoy and Fulé, 2006; Zhang et al., 2006) in order to promote forest resilience and carbon sequestration (Millar et al., 2007). On the other hand, cautions against post-fire planting arise from abundant post-fire natural tree regeneration in some regions (Shatford et al., 2007), evidence that successful dense plantings increase future fire severity (Thompson et al., 2007), and preference for natural recovery processes (Hairston and McGarigal, 2010).

Forests dominated by ponderosa pine (*Pinus ponderosa*, Lawson and C. Lawson) in the southwestern U.S. are a good example of a forest region where the need for post-fire tree planting is unresolved. Tree density in most ponderosa pine forests in the southwestern U.S. is currently greater than conditions documented prior to European settlement (*circa* 1880) due to early clear-cutting of old trees and over-grazing, followed by large region-wide pulses of tree regeneration and the suppression of low-intensity fires (Savage and Swetnam, 1990; Covington and Moore, 1994; Swetnam et al., 1999). Current forests are significantly more prone to large-scale severe burning than they were historically (Allen et al., 2002; Williams et al., 2010).

Severely burned ponderosa pine forests of the southwestern U.S. are at risk of conversion to non-forested ecosystems due to slow, sparse pine regeneration (Savage and Mast, 2005; Dore et al., 2012; Roccaforte et al., 2012; Savage et al., 2013).

Natural regeneration of ponderosa pine following severe wildfire often takes many decades, especially in large openings distant from seed trees (Bonnet et al., 2005; Haire and McGarigal, 2010). For example, ponderosa pine regeneration was considered to be unsuccessful for the first decade after the high severity Cone Fire in northern California based on less than 72 established seedlings ha⁻¹ (Ritchie and Knapp, 2014). In particular, plots with distance to seed trees greater than 60 meters (m) had less than 25 established seedlings ha⁻¹ (Ritchie and Knapp, 2014). A recent study in the southwestern U.S. showed that ponderosa pine did not regenerate within the first decade in 57% of severely burned stands (Roccaforte et al., 2012). A longer term study of two severely burned sites in Arizona and New Mexico reported little natural regeneration of ponderosa pine for the first decade after burning, followed by a slow moving front of regeneration in the second decade that extended 200 m from seed source (Haire and McGarigal, 2010).

Slow regeneration after severe wildfire can be attributed to many factors that limit ponderosa pine seedling establishment. Basic regeneration requirements of ponderosa pine have been well established in the literature (Schubert, 1974; White, 1985) as a large supply of seed; a seedbed with loose soil, light litter cover, and mostly free of competing vegetation; light shade; consistent moisture for seed germination and early seedling growth; and low populations of seed- and seedling-eating mammals, birds, and insects. Slow regeneration of ponderosa pine after severe fire has been attributed to a lack of seed trees and its lack of fire-adapted serotinous cones and vegetative resprouting (Burns and Honkala, 1990; Haire and McGarigal, 2010; Puhlick et al., 2012). Post-fire factors that can make the seedbed unfavorable for ponderosa pine regeneration include high soil temperature (Kasischke and Johnstone,

2005), topsoil erosion (Moody and Martin, 2001), and heavy herbaceous competition (Pearson, 1942). These negative influences on natural seedling establishment may be ameliorated by “nurse logs” or lopped and scattered slash from logging activities that create favorable microsites (Salonius et al., 1996; Castro et al., 2011). Shrubs and herbaceous plants may facilitate seedling establishment initially (Keyes et al., 2005; Keyes et al., 2009; Redmond et al., 2012), but later become resource competitors (Peppin et al., 2010; Huffman et al., 2012). Herbivory from ungulates and burrowing mammals can negatively influence seedling survival and growth (Heidmann, 2008; Waring and Goodrich, 2012). The occurrence of environmental conditions conducive to regeneration of ponderosa pine after high-severity fire is projected to decrease over the next century due to climate changes that reduce water availability to seedlings and increase risk of frost damage (Williams et al., 2013; Savage et al., 2013). Regeneration failures and possible range contraction have recently been reported for ponderosa pine in the western U.S. (Bell et al., 2014).

Planting ponderosa pine after severe wildfire may accelerate reforestation, yet little is known about survival of planted seedlings in recent reforestation projects, or whether planting is even needed. In the southwestern U.S, planting of ponderosa pine after wildfire is increasing after many decades of minimal planting (Sánchez Meador and Graves, 2011). Interest in planting ponderosa pine in the southwestern U.S. likely will increase in the future as high-severity fires, bark beetle outbreaks, and drought-induced tree dieoffs (e.g., Williams et al., 2010; Williams et al., 2013) leave sites without appropriate forest cover required by the National Forest Management Act for federal lands. Moreover, planting can be used to

148 introduce heat- and drought-tolerant populations to deforested sites for the purpose of
149 mitigating climate change impacts (Rehfeldt et al., 2014).

150 Constraints to establishment of planted seedlings are similar to those that influence
151 natural regeneration, with the addition of agency-, manager-, and contractor-influenced
152 variables. Planting practices important to regeneration success include decisions about seed
153 source, seedling stock type, storage, site selection, timing and method of planting, and
154 mitigation of browsing and competing vegetation (Schubert et al., 1970; Heidmann and Haase,
155 1989; Heidmann, 2008; Pinto et al., 2011). Survival of planted ponderosa pine seedlings in
156 recent post-fire reforestation projects is not well known, but can vary widely. For example,
157 survival of ponderosa pine seedlings planted following the Cone Fire in northern California
158 ranged from 30% to 96% among plots ten years after planting (Ritchie and Knapp, 2014).

159 Most research on performance of planted ponderosa pine seedlings has focused on
160 private forestlands in relatively mesic regions in its range (e.g., Pinto et al., 2011; Ritchie and
161 Knapp, 2014). The need for post-fire planting of ponderosa pine in the southwestern U.S. is
162 unknown and has not been widely considered aside from a recent recommendation for
163 managers to facilitate natural post-fire recovery processes and to be patient because natural
164 regeneration is slow after severe burning (Haire and McGarigal, 2010). Our study is the first (to
165 our knowledge) to measure ponderosa pine seedling survival on multiple planted sites after
166 severe wildfires and compare seedling density between planted and adjacent unplanted areas.
167 We addressed the following questions using data collected at eight severely burned sites in
168 Arizona and New Mexico: What is the survival of recent post-fire plantings of ponderosa pine?
169 What tree densities result from planting? What tree densities result from natural post-fire

regeneration? What biotic and abiotic factors are associated with post-fire seedling establishment, and do these factors differ between planted and naturally established seedlings? Is natural regeneration after severe burning adequate to produce desired future forest conditions? Our results provide guidelines for tree planting in post-fire reforestation of ponderosa pine forests in the southwestern U.S.

2. Methods

2.1. Study sites

In summer 2013, we sampled post-fire plantings of ponderosa pine at eight sites that burned between 2002 and 2005 distributed over four national forests in Arizona and New Mexico (Table 1). We selected sites based on availability of information about planting prescriptions and locations and because all had plantings that were at least five years old at the time of our sampling in 2013. Average annual precipitation since burning and since planting was 83% to 88% of the 30-year average for all sites (Table 2). Average annual precipitation in the year of planting was 84 to 102% of the long-term average for all sites.

Across all sites we sampled from a wide variety of soil parent materials, as identified from Terrestrial Ecosystem Survey (TES) map units. At the Borrego Fire (Decimal Degrees (DD) -105.802, 35.981), soil parent materials were granite (four plots), sandstone (five plots), and a combination of these types plus limestone (three plots) (Miller et al., 1993). At the Indian Fire (DD -112.494, 34.501), soil parent material was a residuum from granite for all four plots (Robertson et al., 2000). At the Lakes Fire (DD -106.688, 35.887), soil parent material was sandstone alluvium and tuff (Miller et al., 1993). At the Montoya Fire (DD -106.366, 36.482), soil

parent material was sandstone (six plots) or unclassified (ten plots) (Edwards et al., 1987). At the Pine Canyon Fire, soil parent material was conglomerate sandstone (two plots) or unclassified (six plots) (Edwards et al., 1987). At the Ponil Complex (DD -105.090, 36.680), soil parent material was shale at all plots (Edwards et al., 1987). At the Rodeo-Chediski Fire (DD -110.703, 34.228), soil parent material was alluvium (two plots), sandstone-alluvium (twelve plots), sandstone (five plots), or sandstone-limestone (one plot) (Laing et al., 1987). At the Viveash Fire (DD -105.593, 35.705) soil parent material was granite at all plots (Miller et al., 1993).

Reforestation potential was identified at each site using Robbie (1992), and TES map units of sampled plots are described in Table 3. At the Borrego Fire, we sampled from TES map units 142 (four plots), 155 (one plot), 156 (three plots), 361 (four plots) (Miller et al., 1993), which include several vegetation types. Reforestation potential was rated moderate. At the Indian Fire, one TES map unit (553) was sampled at all four plots where the dominant vegetation type was *P. ponderosa* / *Quercus emoryi* (Torr.) (Robertson et al., 2000) and reforestation potential was low. At the Lakes Fire, TES map units 5 (three plots) and 615 (13 plots) were sampled (Miller et al., 1993). The dominant vegetation type was *Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr. / *Pseudotsuga menziesii* (Mirb.) Franco / *P. ponderosa* for map unit 615 and a grass-dominated type at map unit 5. Reforestation potential was rated low. At the Montoya Fire, TES map units 162 (eight plots), 174 (six plots), and 631 (two plots) were sampled (Edwards et al., 1987). The range of vegetation types is described in Table 3. Reforestation potential was rated low-moderate (two plots), moderate (eight plots), or high (six plots). At the Pine Canyon Fire, TES map units 119 (six plots) and 159 (two plots) were sampled

(Edwards et al., 1987), which were dominated by a *P. edulis* / *Juniperus monosperma* (Engelm.) Sarg. (map unit 119) and *P. edulis* / *J. monosperma* / *Q. gambelii* (Nutt.) (map unit 159) vegetation types. Reforestation potential was rated high (6 plots) and not rated for map unit 159. At the Ponil Complex, TES map unit 264 (Edwards et al., 1987) was sampled, which was dominated by a *P. ponderosa* / *Q. gambelii* vegetation type. Reforestation potential was rated high. At the Rodeo-Chediski Fire, TES map units 178, 182, 187, 191, and 192 (Laing et al., 1987) were sampled, which were dominated by *P. ponderosa*. Reforestation potential was rated low (at five plots), moderate (five plots), or high (at 10 plots). At the Viveash Fire, TES map unit 345 (Miller et al., 1993) was sampled and the dominant vegetation type was *P. ponderosa* / *J. scopulorum* (Sarg.) / *Q. gambelii*. Reforestation potential was rated low for all plots.

2.2. Plot selection

We based sampling intensity at each site on the number of planting units. Planting units were identified as spatially distinct polygons (>2 hectares) where trees were planted based on GIS shapefiles of planting locations provided by National Forest staff. For site-planting year combinations with more than five distinct spatial units (i.e. polygons), each unit was sampled with one 400-m² plot and the number of sampled units was limited to 10. Plot size was designed to sample multiple planted seedlings on the widest spacing used over all sites, and was a compromise among sampling efficiency, large plots needed to adequately sample planted seedlings, and numerous small plots that would be more optimal for detecting seedling presence.

Where there were more than 10 planting units, 10 units were selected using a random number generator. For site-planting year combinations with five or less planting units, two plots per unit were sampled. Each 400 m² circular plot sampled within a planting unit was paired with an unplanted control plot as described below.

We used ArcMap 10 (ESRI, 2011) and either the Spatial Ecology add-on (Beyer, 2004) or Jenness Tool (Jenness, 2012) to create a systematic grid of plot locations across planting units spaced between 50 meters and 200 meters apart, depending on unit size. Tighter grid spacing was used on smaller units and wider grid spacing was used for larger units. To expedite data collection and minimize microsite differences between planted and unplanted plots, the grid was clipped to exclude plots more than 1000 meters from the nearest planting unit boundary. A random number generator was used to select plots from the grid. Paired unplanted plots were selected from the same grid outside the planting unit on the same site. These unplanted plots were selected to have the same burn severity, soil type and vegetation cover type, slope, and aspect as planted plots. Burn severity was obtained from Monitoring Trends in Burn Severity (MTBS) (www.mtbs.gov) data layers, which utilize burned area reflectance classification (BARQ) satellite imagery to compare near- and mid- infrared reflectance values to assess post-fire vegetation condition (USDA and DOI, 2013). A paired t-test on data pooled over all sites showed that planted and unplanted plots had similar slope, aspect, elevation, and burn severity (p value > 0.05).

2.3. Field data collection

We used circular plots of 400 m² at all sites to sample seedlings, vegetation, and other site characteristics. For each ponderosa pine seedling in each plot, condition [live or dead], number of whorls, presence/absence of a “nurse” object within 40 cm, type of nurse object (wood, rock, or live vegetation), direction to nurse object (N, NE, E, etc.), and nurse object size class [small (7.62 – 15.24 cm diameter), medium (15.25 – 30.48 cm diameter), or large (>30.49 cm diameter)] were recorded. Nurse objects were defined as abiotic or biotic features that have the potential to ameliorate microclimatic conditions (Castro et al., 2011). If more than one nurse object was present, up to two were recorded per seedling.

To assess potential sources of competition in each plot, we went from plot center out 5.6 m in each of the four cardinal directions to place a 1 m² vegetation quadrat (Daubenmire, 1959). Each quadrat included measurements of percent aerial and basal cover in categories: forbs, grasses, shrubs, conifer tree seedlings, other trees (deciduous or conifers > 1.37m tall), bole (of standing live trees), bare ground, rock, coarse woody debris (> 7.62 cm), and litter. At 20 locations placed on a consistent, evenly-spaced grid on the plot, canopy cover of woody species (shrubs and trees) was measured using a vertical densitometer 15 cm from the ground level and where canopy cover occurred, and the species/type was recorded. The 15 cm height for canopy cover measurements was chosen to assess plant cover that would provide competition for a tree seedling. On a planar transect from the north end of the plot to the south (22.57m length), a transect (Brown, 1974) was used to measure downed woody debris (> 7.62 cm) by diameter, class (sound/rotten), length, and location. Tree basal area (m² ha⁻¹) was measured for trees > 1.37 m in height with a variable-radius overstory point at plot center using

a 10-basal-area factor prism. Also, distance from plot center to seed trees was recorded up to ~36 m away or 1.5 times the height of the seed tree (Waring and Goodrich, 2012) and height of each seed tree was measured using a laser hypsometer. Seed trees were defined as those that had evidence of recent cone production or were large enough (>40cm diameter breast height) to likely produce cones. Aspect values were transformed into Northness and Eastness values (Palmer, 2014) based on the formula: northness = $\sin(\text{aspect})$ and eastness = $\cos(\text{aspect})$. Both values range from -1 to 1, with 1 being due north or due east, respectively.

2.4. Statistical analysis

We converted seedling counts per plot to seedling density per hectare. A categorical variable was created for seedling presence/absence in each plot and the percentage of seedlings near a nurse object was calculated. Current (2013) plot-level seedling density at planted sites was used to calculate seedling survival based on reported planting density for each site (Table 1).

To test for difference in seedling density between planted plots and unplanted plots, we used a general linear model (GLM) with planting treatment (fixed effect), site (random effect), and the interaction between site and treatment as factors. We considered plot pairing by using a paired Wilcoxon signed-rank test to compare seedling density between planted and unplanted plots on data pooled over all sites. Between seedling density and environmental and vegetation variables were investigated, but these models had high residual deviance and low R^2 (< 0.001), and are not subsequently reported. All statistical analyses ($\alpha = 0.05$) were run in the statistical software package R using the NLME package (R Development Core Team, 2013).

Binomial GLMs that included plot pairs were used to evaluate the effects of planting and other independent variables on seedling presence. To identify biotic and abiotic factors associated with seedling presence, we first selected candidate variables by stepwise regression and then tested them in multivariate GLMs. Candidate variables included plot-level vegetation data and site characteristics, ground cover data, CWD, presence of and distance to nurse objects and seed trees, years since burning, and climate variables. We avoided using intercorrelated variables (p value < 0.05) in the models. To select variables, we conducted stepwise regression models (add 1) in R with seedling presence (1) and absence (0) as responses and with all candidate independent variables, in addition to planting treatment. We used the selected variables in a binomial GLM with random effects for plot pairs, which were nested within unit, which were nested within site. We ran these models on data pooled over both planted plots and unplanted plots, for planted plots only, and for unplanted plots only.

3. Results

3.1. Post-fire vegetation characteristics

Post-fire vegetation characteristics of each site are summarized in Table 4. Over all sites, basal area of overstory ponderosa pine was no more than $2.1 \text{ m}^2 \text{ ha}^{-1}$, and basal area of overstory tree species was no more than $3.6 \text{ m}^2 \text{ ha}^{-1}$. Average canopy cover by shrubs ranged between 1.7% (Ponil Fire) and 63.8% (Indian Fire). Litter cover ranged between 17% (Pine Canyon Fire) and 87% (Viveash Fire). Grass cover ranged between 0% (Indian Fire) and 7% (Pine Canyon Fire). Forb cover ranged between 0% (Indian Fire) and 4.6% (Borrego Fire). Course woody debris ranged between 17 metric tons ha^{-1} (Indian Fire) and 86 metric tons ha^{-1} (Viveash

Fire).

3.2. Effect of planting on seedling presence and density and factors associated with seedling presence

Planting had a significant positive effect ($p < 0.001$) on seedling presence in a GLM model on data pooled across all plots (Table 5). A paired Wilcoxon signed-rank test showed that planted plots had greater seedling density than unplanted plots ($p < 0.001$) for pooled data across all sites. A significant interaction between planting treatment and site in the GLM model for seedling density ($p < 0.001$) showed that the effect of planting differed among sites. Two sites, Indian and Pine Canyon, had no seedlings in planted or unplanted plots (Fig. 1). Of the six sites with seedlings, seedling density was greater in planted than unplanted plots at five sites (Fig. 1). The exception was the Borrego Fire, which had almost three times higher seedling density in unplanted plots than planted plots, and planted plots had two times the average density of planted plots at other sites.

For data pooled over planted plots and unplanted plots, seedling presence was positively associated with planting treatment ($p < 0.001$), years since fire ($p = 0.0333$), and forb cover ($p = 0.009$). For planted plots, seedling presence was positively associated with forb cover ($p = 0.0089$) and litter cover ($p = 0.0297$). For unplanted plots, seedling presence was positively associated with litter cover ($p = 0.0491$) (Table 5).

3.3 Site regeneration and survival of planted seedlings

Seedling density ranged between 0 and 7750 seedlings ha⁻¹ across all plots and all sites, between 0 and 520 (SE 211) on planted sites, and between 0 and 1433 (SE 1270) on unplanted sites. Average whorl counts ranged between 7.8 and 11.8 on sites with seedlings and the average number of seedlings with nurse objects ranged between 21 and 79% (Table 6).

Seedling survival based on our counts in 2013 and manager-reported planting density ranged from 0 to 70% among sites (Fig. 2). Average survival across all sites was 25% (SE 8). Survival was 0% at the Pine Canyon and Indian Fires and ranged between 12 and 70% among the other six sites (Fig. 2). Average site-level seedling survival (n=8) was significantly and positively related to forb cover ($r^2 = 0.77$; $p < 0.001$), whereas all other relationships with site-level physical and vegetation characteristics were weak and not significant.

4. Discussion

Our results suggest that half of post-fire reforestation projects in Arizona and New Mexico that planted ponderosa pine seedlings were successful at establishing enough seedlings to likely produce target amounts of mature trees. Results of the generalized linear models supported the paired Wilcoxon signed-rank test results by showing a significant effect of planting on seedling density. The other half of projects failed to establish enough seedlings, or in one case, planted seedlings on a site that eventually had ample natural regeneration. At most sites, however, natural regeneration during the first decade after burning did not produce enough seedlings. Below we explain and discuss our results and conclusions in more detail.

Whorl counts, which we used as a proxy for seedling age (Haire and McGarigal, 2010), showed that the vast majority of seedlings at all sites established after burning. It is possible

that seedlings in planted plots originated both from planting and natural regeneration. Distinguishing planted from naturally regenerated trees in burned areas can be difficult (Savage and Mast, 2005). Seedling age based on whorl counts in planted plots was within one year of manager-reported planting years for the Borrego, Lake, and Viveash Fires, and was 1-3 years older than reported planting years for the Montoya, Ponil, and Rodeo-Chediski Fires. This difference between whorl counts and planting year is expected given that seedlings were one or two years old at the time of planting, and because whorl counts provide an approximate estimation of seedling age.

We found similarity and difference in biotic and abiotic factors associated with post-fire seedling establishment between planted and unplanted plots. Time since fire was positively associated with seedling presence for all plots. Litter cover was positively associated with seedling presence in planted and unplanted plots. This result is consistent with earlier reports of positive effects from scorched needles or dead grass on soil moisture and seedling recruitment (Schubert et al., 1970; Pannkuk and Robichaud, 2003; Bonnet et al., 2005; Heidmann, 2008). Litter and other non-competitive ground covers can ameliorate the severe temperature fluctuations that occur in the post-fire environment, reduce frost-heaving, and improve moisture retention (Heidmann, 2008). We discovered a positive association between forb cover and seedling presence for planted plots, but not for unplanted plots. A positive influence of forb cover on seedling presence is inconsistent with earlier research that reported interference with establishment of ponderosa pine seedlings by herbaceous plants (Schubert et al., 1970; Bonnet et al., 2005). At our study sites where forb cover was low (<5 %), forbs appeared to facilitate rather than impede the establishment of planted seedlings. Negative

relationships between ponderosa pine seedling occurrence and herbaceous cover have been reported at higher cover (>10%) than occurred in our study (Bonnet et al., 2005). Thus, it is likely that the amount of herbaceous cover on our sites did not surpass a competitive threshold. Another interpretation of our results is that site conditions conducive to tree seedling establishment also were conducive to forb establishment and recovery after burning.

Our results show variable success in establishing ponderosa pine by planting into severely burned sites in the southwestern U.S. between 2004 and 2008. Survival of planted seedlings varied between 0% and 70% and averaged 25% (SE 8) among the eight sites we investigated. This result is based on seedling density measured in 2013 (five to eight years after planting) and manager-reported planting density across all planting units on a given site. Actual planted density may have been lower than reported density due to skipping over unplantable areas. Furthermore, plot size and number of plots may have limited our ability to accurately measure survival. Nonetheless, the range of seedling survival we measured over sites (0 to 70%) is plausible considering that some plantings are highly successful and others are complete failures. Our estimated survivorship of ponderosa pine seedlings planted after severe wildfire in Arizona and New Mexico for the period of five to eight years after planting is lower than reported for the Cone Fire in northern California (30 to 96% over all plots) ten years after planting (Ritchie and Knapp, 2014). This comparison suggests lower survival of planted ponderosa pine seedlings in Arizona and New Mexico than in other regions. Seedling survival in our study may have been reduced by drought, as precipitation was 83 to 88% of 30-year-normals when averaged across all years since burning.

Our results are a caution to forest managers that establishing ponderosa pine by planting after severe fires in the southwestern U.S. is a challenge and that failures are to be expected. Three of eight, or 38% of sites, had 0-12% survival of planted seedlings, and, of those, two had 0% survival. We suspect that harsh site conditions played a role in the total failure of plantings at the Indian and Pine Canyon Fires. The Pine Canyon planting units we surveyed were located in a transition zone between ponderosa pine forest and drier piñon-juniper woodland. At the Indian Fire, sprouting woody species averaged 60% cover and may have provided intense competition with planted seedlings. Also, soil at the Indian Fire was a well-drained granitic residuum and coarse sandy loam, which may have reduced soil water available to seedlings, and reforestation potential was rated low (Robbie, 1992). The Viveash Fire, where seedling survival was only 12%, also had granite-derived soil and reforestation potential rated as low. Future research about survival of planted seedlings in the southwestern U.S. should investigate factors that we did not address due to lack of data, such as seed source, seedling stock type, storage conditions and duration, planting site selection, and salvage logging.

Natural regeneration by ponderosa pine at the severely burned sites in our study was less successful and more variable over sites than artificial regeneration. Seedling density produced by natural regeneration in the seven to ten years after burning was zero at two sites, and less than 50 trees ha⁻¹ at five of the remaining six sites. In contrast, natural regeneration produced about 1,500 trees ha⁻¹ at one site (Borrego). Our finding of variable natural regeneration of ponderosa pine after severe burning is consistent with earlier reports (Bonnett et al., 2005; Savage and Mast, 2005; Haire and McGarigal, 2010; Roccaforte et al., 2012). Natural regeneration of ponderosa pine in burned areas typically requires over one decade

when seed trees are 100 m or more away (Bonnett et al., 2005; Haire and McGarigal, 2010). Heavy natural regeneration of ponderosa pine within one decade after burning, which we documented only at the Borrego Fire, is uncommon based on our measurements at eight burned sites. Prolific natural regeneration at the Borrego Fire likely resulted from high seed production and dispersal by trees that survived the fire, and favorable conditions for seedling establishment, which require further investigation.

Our finding that ponderosa pine seedling presence increased with time since burning is consistent with other studies of post-fire conifer regeneration (Cormier et al., 1990; Haire and McGarigal, 2010; Huffman et al., 2012), and suggests that seedling densities at our study sites may increase in the future due to further natural regeneration. Yet, climate warming and increasing aridity in the southwestern U.S. may constrain further natural regeneration. Climate-envelope models predict a contraction of ponderosa pine's range in the southwestern U.S. over the next century (Rehfeldt et al., 2014). Future climate warming combined with severe burning is expected to narrow the occurrence of environmental conditions conducive to ponderosa pine regeneration (Feddema et al., 2013; Savage et al., 2013). Natural regeneration of ponderosa pine is sparse in drier parts of its range in the southwestern U.S. (Puhlick et al., 2012) and the entire region is expected to become more arid (Seager et al., 2007; Williams et al., 2013). Thus, the amount of natural regeneration of ponderosa pine we documented after severe burning for the period of 2004 to 2008 may be rarer in the future.

Given that many current forest health problems in ponderosa pine forests of the southwestern U.S. result from a high density of trees that established in the early 1900s (Savage and Swetnam, 1990; Covington et al., 1997; Allen et al., 2002), seedling densities in stands

recovering from severe burning should be carefully considered. Too few seedlings may result in long-term conversion of ponderosa pine forests to grass- or shrub-lands (Savage and Mast, 2005; Roccaforte et al., 2012). In contrast, too many seedlings could produce overly dense stands and heavy fuels that are partly responsible for current widespread intense wildfires and other forest health problems.

Contemporary forest restoration projects in southwestern ponderosa pine forests prescribe 56 to 138 mature trees ha^{-1} based on historical range of variation and restoration ecology principles (Reynolds et al., 2013). Survival of ponderosa pine seedlings from the 5th to the 50th year after planting averaged 44% at the Fort Valley Experimental Forest in northern Arizona (DeWald and Mahalovich, 2008). Based on this survival percent, four of eight sites in our study (Lakes, Montoya, Ponil, Viveash) have a seedling density in planted plots (125 to 240 ha^{-1}) that will produce a density of mature trees (55 to 106 ha^{-1}) close to the target density.

In contrast, planting at three sites did not establish enough seedlings to produce desired density of mature trees. Current density of planted seedlings in the areas we sampled at the Rodeo-Chediski Fire is expected to produce a density of mature trees (44 ha^{-1}) 21% below the desired target. Total mortality of all planted seedlings and absence of natural regeneration at the Indian and Pine Canyon Fires strongly suggest they will be deficient in mature trees in the foreseeable future.

Natural regeneration in unplanted plots during the first decade after burning in our study did not produce a range of seedling density consistent with desired numbers of mature trees at any site. Two sites (Indian, Pine Canyon) had no naturally established seedlings, and five sites (Lakes, Montoya, Ponil, Rodeo-Chediski, Viveash) had seedling density (3 to 33 ha^{-1})

that is currently too low to produce the target number of mature trees, assuming 44% survive to year 50. Current density of planted (520 ha^{-1}) and naturally established ($1,434 \text{ ha}^{-1}$) seedlings at the Borrego Fire will produce an overstocked stand, with 229 mature trees ha^{-1} expected in planted plots and 631 ha^{-1} expected in unplanted plots. High density of ponderosa pine seedlings, produced by abundant natural regeneration or overplanting, will require subsequent thinning or prescribed burning to reduce tree density, fuels, and risk of another severe wildfire (Agee and Skinner, 2005; Finney et al., 2006; Fulé et al., 2012; Shive et al., 2013).

In summary, we found wide variation in survival of ponderosa pine seedlings planted at eight sites in Arizona and New Mexico that had been severely burned between 2002 and 2005. Seedling density in planted plots was consistent with the desired target density of mature trees based on historical range of variability and restoration ecology principles at four of eight sites (50%), below the target at three sites, and above the target at one site where natural regeneration was prolific. Naturally regenerated seedlings occurred at six of eight sites (75%) within the first decade after burning, but current density at all but one site is below amounts needed to produce the target density of mature trees. Predictions of increasing forest fires and climate warming (Liu et al., 2010; Flannigan et al., 2013; Williams et al., 2013; Xu et al., 2013) emphasize the importance of directing resources for tree planting to severely burned sites that have little potential for natural seedling establishment, yet acceptable potential for survival of planted seedlings. Future research should build on our findings with investigations that increase understanding of the numbers of established seedlings needed to produce desired density and spatial pattern of mature trees, as well as site and fire characteristics and management actions that promote natural regeneration and increase survival of planted seedlings.

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728 **Tables:**

729 Table 1. Planting projects surveyed in Arizona and New Mexico by National Forest, site (year
 730 burned), fire size, year planted (season if known), total hectares (HA) planted, planting density
 731 in trees per hectare (TPH) from manager surveys, number of plots and units sampled per site
 732 (PL is planted, UNPL is unplanted), average elevation, and burn severity. Unk. = unknown. Burn
 733 severity derived from Monitoring Trends in Burn Severity data (USDA and DOI, 2013).

National Forest	Site (Year Burned)	Size (ha)	Year Planted	Total HA Planted	Planting Density (TPH)	No. of Sampled Plots (Units)	Elevation (m)	Burn Severity
Apache-Sitgreaves	Rodeo-Chediski (RC) (2002)	186,953	2008	294	234	10 PL / 10 UNPL (10 units)	2056	4 – 5
	Viveash (VI) (2000)	9,097	2004	55	1074	2 PL / 2 UNPL (1 unit)	2745	3 – 4
Santa Fe	Borrego (BG) (2002)	5,213	2005	107	748 - 1074	6 PL / 6 UNPL (6 units)	2600	3 – 4
	Lakes (LK) (2002)	1,743	2005	192	479	8 PL / 8 UNPL (4 units)	2534	2, 4
Carson	Ponil Complex (PO) (2002)	36,066	2005	259	741	6 PL / 6 UNPL (3 units)	2460	3 – 4
	Pine Canyon (PC)	1,656	2008	162	284	4 PL / 4 UNPL (2	2320	2 – 3

	(2005)					units)		
	Montoya (MO) (2002)	1,666	2006	241	746	8 PL / 8 UNPL (8 units)	2621	2 – 4
Prescott	Indian (IN) (2002)	642	2005	130	1356	2 PL / 2 UNPL (1 unit)	1811	3 – 4

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736 Table 2. Average annual precipitation as a percent of the 30-year-average by site and year
737 (PRISM 2013). Year of burning is in **bold** text; year of planting is in **bold italicized** text. Site
738 codes are defined in Table 1.

Site	Year												
	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012
BG	0.88	0.77	0.65	0.74	0.94	0.93	0.92	0.97	0.94	0.79	0.96	0.76	0.67
IN	0.81	0.82	0.70	0.75	0.85	0.99	0.95	1.04	0.94	0.83	0.84	0.80	0.59
LK	0.87	0.78	0.65	0.75	0.93	0.92	0.92	1.01	0.94	0.79	0.96	0.78	0.70
MO	0.84	0.80	0.71	0.75	1.01	0.99	0.97	1.02	0.96	0.84	0.88	0.75	0.63
PC	0.82	0.72	0.64	0.67	0.86	0.88	0.86	0.92	0.84	0.75	0.77	0.68	0.55
PO	0.83	0.80	0.73	0.76	1.06	0.98	0.98	1.04	0.94	0.85	0.87	0.74	0.62
RC	0.90	0.80	0.55	0.82	0.92	0.88	0.81	0.98	1.02	0.56	1.09	0.83	0.69
VI	0.86	0.77	0.68	0.69	0.92	0.90	0.97	1.06	0.95	0.84	0.94	0.78	0.66

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740 Table 3. Woody species by terrestrial ecosystem survey (TES) map unit for plots sampled at four
741 national forests in Arizona and New Mexico.

Forest*	TES Map Unit No.	Dominant species [†]
Apache-Sitgreaves	178	PIPOS/PIED/JUDE/JUMO/QUGA
	182	PIPOS/JUDE/QUGA
	187	PIPOS/PIED/JUDE/JUMO/QUGA
	191	PIPOS/QUGA
	192	PIPOS/QUGA
Carson	119	PIED/JUMO/QUGA
	159	PIED/JUMO
	162	PIPOS/PIED/QUGA
	174	PIPO/QUGA
	264	PIPO/QUGA
	631	QUGA/SASC/PIPO

Prescott	553	PIPOS/QUEM
Santa Fe	5	POPR/FEAR
	142	ABCO/PSME/PIPOS/POTR/QUGA
	155	PIPOS
	156	PIPOS
	345	PIPOS
	361	PIPOS/PIED
	615	ABCO/PSME/PIPOS/POTR/QUGA

*Data were derived from forest-specific soils reports.

† Woody species codes: PIPOS = *Pinus ponderosa* var. *scopulorum*, PIED = *Pinus edulis*, JUDE = *Juniperus deppeana*, JUMO = *Juniperus monosperma*, QUGA = *Quercus gambelii*, QUEM = *Quercus emoryi*, ABCO = *Abies concolor*, JUSC = *Juniperus communis*, POTR = *Populus tremuloides*, PSME = *Pseudotsuga menziesii*

Table 4. Average values (standard error in parentheses) of vegetative characteristics for eight burned sites in Arizona and New Mexico pooled over planted and unplanted plots.

Site	BA PIPO*	BA all species*	Canopy Cover [†]	Litter Cover [‡]	Grass Cover [‡]	Forb Cover [‡]	Avg. CWD (metric tons ha ⁻¹) [§]
	(m ² ha ⁻¹).....		(%).....				
Borrogo	1 (0.7)	2.1 (1.2)	26.7 (5.4)	76 (4.4)	1.1 (0.3)	4.6 (1.0)	39.3 (11)
Indian	1.7 (1.7)	1.7 (1.7)	63.8 (4.7)	40 (6.9)	0 (0)	0 (0)	29 (14.2)
Lakes	2.1 (1.5)	3.6 (2.2)	27.2 (6.6)	50 (5.0)	1.5 (0.4)	1.7 (0.3)	46.6 (8.2)
Montoya	1 (0.8)	1 (0.8)	48.8 (4.4)	64 (5.1)	4.0 (0.7)	3.9 (1.0)	17.3 (6.9)
Pine Canyon	0 (0)	0 (0)	5.5 (2.9)	17 (3.7)	6.9 (1.7)	1.5 (0.3)	33.5 (24.6)
Ponil	0 (0)	0 (0)	1.7 (0.9)	26 (5.1)	5.8 (0.8)	2.3 (0.4)	73.2 (11.1)
Rodeo-Chediski	0.1 (0.05)	0.1 (0.05)	11.0 (3.8)	57.5 (4.5)	3.1 (0.6)	3.5 (0.5)	23.2 (4.3)
Viveash	0 (0)	0.6 (0.6)	21.3 (8.3)	87.4 (0.9)	1.9 (0.4)	0.8 (0.2)	85.7 (21.1)

*Average basal area (BA) (restricted to *Pinus ponderosa* [PIPO] and separately for all species); †canopy cover by tree and shrub species (CC) measured using a densitometer; ‡percent cover of litter, grass, and forb from Daubenmire quadrats; § coarse woody debris (CWD). Variables with minimal variation or low presence, and duplicative measures were excluded (e.g. CWD, shrub cover, rock, bare ground, bole, and other tree cover from Daubenmire quadrats).

Table 5. Results of generalized linear mixed models of plot-level seedling presence across eight burned areas in Arizona and New Mexico.

Data source	Significant variable	Coefficient	p value	RD (LL)*	df [†]	N [‡]	Conditional R ²
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All	Planting	3.078	< 0.001	83	n/a	92	0.74
	Years since fire	1.672	0.0333	(-42)			
	Percent forb cover	0.4469	0.0090				
Planted	Percent forb cover	0.9638	0.0089	37	43	46	0.69
	Percent litter	0.0376	0.0297				
Unplanted	Percent litter	0.026	0.0491	54	44	46	0.13

*RD is Residual Deviance and LL is Log Likelihood. [†]Df is degrees of freedom. [‡]N is sample size. The sample size is smaller than Table 1 numbers due to issues with missing data.

Site	Min	Max	Average Planted	Average Unplanted	Whorl Counts	Percent with Nurse Object
	(seedling density, # ha ⁻¹).....					
Borrogo	0	7750	520 (211)	1433 (1270)	7.8 (0.1)	45 (10)
Indian	0	0	0	0	n/a	n/a
Lakes	0	425	156 (59)	12.5 (9.4)	7.8 (0.09)	71 (9)
Montoya	0	500	241 (55)	3.1 (3.1)	10.4 (0.1)	66 (10)
Pine Canyon	0	0	0	0	n/a	n/a
Ponil	0	250	146 (38)	29 (25)	11.8 (0.08)	79 (8)
Rodeo- Chediski	0	200	98 (21)	35 (18)	6.6 (0.9)	55 (9)
Viveash	0	175	125 (50)	12.5 (12.5)	9 (0.1)	21 (11)

754 **Figures:**

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Figure 1. Average ponderosa pine seedling density (SE shown as vertical bars) for planted and unplanted plots at eight burned sites in Arizona and New Mexico. Scale is consistent for all sites except Borrego.

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Figure 2. Percent survival of planted ponderosa pine seedlings on eight burned sites in New Mexico and Arizona. Number of 400 m² plots sampled is in parentheses below site name.