EFFECTS AND EFFECTIVENESS OF SEEDING FOLLOWING HIGH-SEVERITY WILDFIRES IN NORTHERN ARIZONA PONDEROSA PINE FORESTS

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ABSTRACT

EFFECTS AND EFFECTIVENESS OF SEEDING FOLLOWING

HIGH-SEVERITY WILDFIRES IN

ARIZONA PONDEROSA PINE FORESTS

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Seeding following high-severity wildfires is intended to reduce water runoff, soil erosion, and non-native species invasions. Post-wildfire seeding treatments most commonly use seed of non-native annual cereal grains selected for quick growth and abundant root systems. U.S. policy recommends the use of native seed, but adequate supplies of locally collected seed are rarely available. Seeding following wildfires for erosion control is based on the positive correlation between bare ground and increased sedimentation rates, and the negative correlation between vegetative (or ground) cover and runoff and erosion rates. Seeding to prevent non-native species invasions is based on the concept that seeded species will quickly co-opt resources and exclude non-natives. Critics of seeding following wildfires argue that seeding rarely achieves the cover levels needed to effectively reduce runoff and erosion; when treatments do achieve high levels of cover, it comes at the expense of native plant regeneration. Due to the stochastic nature of wildfires, well-replicated studies with adequate control areas have been rare. I used a controlled, replicated, and randomized experimental design to test the effects and effectiveness of native and non-native seeding following three high-severity wildfires in Arizona ponderosa pine (*Pinus ponderosa* C. Laws.) forests. I found seeding with either non-natives or native species: (1) was generally ineffective in producing the levels of vegetative cover associated with reduced runoff and erosion; (2) significantly altered

plant community composition; and (3) did not reduce abundance of non-native species. My results add to the growing evidence that seeding following high-severity wildfires does not achieve management objectives of providing high vegetative cover associated with low soil loss and suppression of non-native species. Seeding also altered post-fire plant community composition through reductions in native species, including perennial native forbs, shrubs, and colonizing annual/biennial species. The high financial cost and low potential for effectiveness should call into question the continued practice of seeding areas burned in high-severity wildfires.

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PREFACE

This thesis contains chapters intended for publication and is written in manuscript format with the literature cited placed at the end of each chapter. Chapter 3 and 4 use “we” instead of “I” because these papers have co-authors.

CHAPTER ONE

Introduction

Land managers have applied seed to areas burned by high-severity wildfires for nearly a century (Beyers 2004). The intention of post-fire seeding has shifted over time. Early efforts were focused on increasing forage and browse for cattle and wildlife species (Beyers 2004). More recent efforts are focused on promoting native species regeneration and reducing water runoff and soil erosion (Robichaud et al. 2000, Beyers 2004). Presently, the rationale for seeding following fires as stated in federal policy is: "determine the need for and prescribe and implement emergency treatments to minimize threats to life or property or to stabilize and prevent further unacceptable degradation of natural and cultural resources resulting from the effects of a fire" (USDA 2004). Federal policy states the purpose of post-fire seeding is to reduce runoff and soil erosion and prevent invasion by non-native species (USDA 2004).

Post-fire seeding is based on the expectation that high amounts of vegetative cover will result in increased water infiltration into the soil (Debano et al. 1998, Pannkuk and Robichaud 2003). Conceptually, seeded species are expected to grow rapidly and provide abundant vegetative cover which will directly protect soil from rainsplash erosion (Robichaud et al. 2000). The roots of seeded species are intended to aid in the removal of hydrophobic soil layers that may form as a result of fire and to encourage water infiltration into the soil surface (Debano et al. 1998, Robichaud et al. 2000).

Species selected for post-fire seeding treatments have shifted over time but have generally been quick-growing species intended to provide high amounts of above-ground vegetative cover and below-ground biomass (Beschta et al. 2003, Beyers 2004). Over the

past few decades, seeds of non-native annual cereal grains or sterile graminoid hybrids have been most commonly utilized (Beyers 2004). United States (U.S.) federal policy encourages the use of native species when they are available (USDA 2004). However, inclusion of native species in post-fire seed mixes is rare due to unavailability of locally collected seed stock and high costs (Beschta et al. 2003). Several studies have reported that when native species are included in post-fire seed mixes, the natives are typically not from locally occurring populations, but instead originate from hybridized field-grown sources (Robichaud et al. 2000, Barclay et al. 2004, Hunter et al. 2006, Kuenzi et al. 2008). Most commonly, post-fire native seed mixes include native seed and non-native cereal grains seeded together (Barclay et al. 2004, Hunter et al. 2006, Kuenzi et al. 2008). This practice is controversial because seed mixes that include natives and cereal grains have been shown to reduce native species success and seeding with native hybrids may have long-term negative effects on the genetic make-up of locally adapted species (Millar and Libby 1989, Zelnik et al. 2008).

Previous post-wildfire seeding research has indicated low success rates of seeding and frequent negative ecological impacts to native plant regeneration (Robichaud et al. 2000, Beyers 2004). Forested ecosystems generally have very low values of exposed bare ground with as much as 80% of the ground covered by vegetation, litter, and duff with correspondingly low runoff and erosion rates (Debano et al. 1998, Johansen et al. 2001). Hydrological studies indicate runoff and erosion rates are reduced to background levels when ground cover increases above 60% and that the risk of increased runoff and erosion is greatest in the first growing season following fire (Benavides-Solorio and MacDonald 2001, Johansen et al. 2001, Pannkuk and Robichaud 2003). These studies

have demonstrated variability in the amount of ground cover required to reduce runoff and erosion; the range from forested ecosystems is from 40-70% (Wagenbrenner et al. 2006). Studies report consistent and significant decreases in runoff and erosion when bare ground is <60% and vegetative cover is >60%, and these are commonly-used target levels employed by researchers and land managers (Robichaud et al. 2000, Beyers 2004, Keeley 2004). Many post-fire seeding studies from forested ecosystems report vegetative cover values in the first growing season following fire of <30% (Robichaud et al. 2000, Barclay et al. 2004, Beyers 2004, Hunter et al. 2006, Wagenbrenner et al. 2006, Robichaud et al. 2006). Those studies that report higher levels of vegetative cover from seeded species commonly report negative impacts to native plant regeneration (Amaranthus et al. 1993, Keeley 2004). These impacts include reduced species richness, reduced regeneration of post-fire endemic species, and reductions or exclusion of long-lived perennials, shrubs, and conifers important for long-term soil stability and ecological functioning (Schoennagel and Waller 1999, Keeley 2004, Kruse et al. 2004).

Plant community composition and species richness are influenced by high dominance of one or a few species in an environment, and seeding may increase competition for resources in post-fire environments (Grace 1999, Grime 2001, Smith et al. 2004). The quick growth attributes that make seeded species desirable from the perspective of establishing abundant and rapid vegetative cover are correlated with rapid utilization of resources (Hunter and Omi 2006). Seeded non-native species can utilize soil moisture and nutrients at rapid rates and potentially gain a competitive advantage over native species (Amaranthus et al. 1993, Hunter and Omi 2006). High dominance of one or a few species in an ecosystem may limit recruitment and decrease species richness

(Smith et al. 2004). Several studies have demonstrated that when seeded species dominate in terms of high cover relative to all other species, species richness and diversity are decreased (Conard et al. 1991, Amaranthus et al. 1999, Barclay et al. 2004, Keeley 2004).

Due to the stochastic nature of wildfires, post-fire seeding studies which utilize a controlled, replicated, and randomized experimental design are rare (Robichaud et al. 2000). Most post-wildfire seeding studies have been conducted in an opportunistic fashion in that they observe the effects of seeding treatments applied by land management agencies. Large-scale seeding treatments are commonly applied via aerial applications. This is appropriate for landscape-scale treatments, but can create imprecision in research, in that the amount of seed applied to a given study plot is variable. Researchers have also indicated difficulty in locating unseeded areas to use as control areas (Schoennagel and Waller 1999, Barclay et al. 2004).

In this study, I took advantage of three separate, high-severity wildfires to quantify the effects and effectiveness of native and non-native seeding in the first and second years following high-severity wildfires in ponderosa pine (*Pinus ponderosa* C. Lawson) forests using controlled, replicated, and randomized experimental seeding treatments**.** The primary goal of the analyses of the first year data was to determine if seeding was effective at creating high vegetative cover, reducing bare ground, or reducing non-native species abundance. I then used data from the first and second year following fire, collected from two fires, to examine if seeding affected post-fire plant community regeneration, including composition and abundance of life-form groups and individual plant species.

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CHAPTER TWO

Literature Review

Wildfires are a recurring and important component of ecosystems across the western United States; but alterations caused by Euro-American settlement and subsequent management practices have negatively impacted forest structure and functioning (Agee 1993, Fulé et al. 1997). Fires are integral in maintaining ecosystem attributes including forest vegetative structure, composition, and nutrient cycling (DeBano et al. 1998). In many western U.S. forests, fire suppression, livestock grazing, and logging have altered forest structure and composition, and thus presettlement fire regimes (Agee 1993, Covington and Moore 1994, Fulé et al. 1997). Fire regimes of forest types with long fire return intervals and characteristically high-severity fire, such as some mixed-conifer and spruce-fir forests, may be relatively unaffected by a century of fire suppression (Agee 1993, Fulé et al. 2003). In contrast, forests with historical fire regimes that included short fire return intervals typified by low-severity fire may be the most structurally altered and also at high risk for stand-replacing fires (Covington and Moore 1994, Fulé et al. 1997).

Increases in the extent and severity of wildfires in some western forests are also linked with drought cycles and climate fluctuations (Agee 1993, McKenzie et al. 2004, Westerling 2006). Long-term studies on forest dynamics show an increase in tree mortality across the western United States and indicate climate fluctuations as a potential driver of the change (Van Mantgem et al. 2009). Climate modeling suggests that some forests, particularly those in the southwestern U.S., will continue to experience lower precipitation that will further affect future fire regimes (Seager et al. 2007). Thus, with a

legacy of management actions that altered ecosystem attributes in some forests, climate change, and increasing tree mortality, it is likely that high-severity wildfires will continue to occur in the western U.S.

The shift from frequent low intensity surface fires to large scale high-severity wildfires in southwestern ponderosa pine (*Pinus ponderosa* C. Laws.) forests may have long-term effects on the structure and composition of plant communities (Crawford et al. 2001, Keeley et al. 2003, Savage and Mast 2005). Historic fire regimes were characterized by low severity frequent fires with return intervals of between 2-20 years; but presently return intervals and severity are greater than the historic reference conditions (Baisan and Swetnam 1990, Fulé et al. 1997). Savage and Mast (2005) noted two discrete possible trajectories for ponderosa pine forests that experienced high-severity wildfires. The first involved dense regeneration of ponderosa pine trees leading to the continued high potential for recurring high-severity fires; the second was a lack of recruitment of conifers and establishment of persistent grass and shrubland communities (Savage and Mast 2005). While the ultimate fate of plant communities following high-severity wildfires in southwestern ponderosa pine forests is uncertain, it is clear that the immediate effects often include altered plant community composition and changes to ecosystem functioning (Schoennagel and Waller 1999, Crawford et al. 2001, Keeley et al. 2003, Laughlin et al. 2004).

Undisturbed forested ecosystems have high proportions of ground covered by vegetation and litter, resulting in very low runoff and erosion rates (Debano et al. 1998). Commonly pre-fire ground cover exceeds 80% in many forest types (Robichaud et al. 2000). By definition, high-severity wildfires remove or consume large percentages of

standing crop, litter, and duff layers (Ryan and Noste 1985). This reduction in ground cover can lead to a decrease in water infiltration into the soil, increase rain splash erosion, and contribute to increases in sheet and overland flows, all of which can result in high sediment yields and erosion (DeBano et al. 1998, MacDonald and Huffman 2004). Severe erosion can negatively impact water quality, riparian species, instream species, and potentially affect long-term ecosystem productivity and biodiversity (Robichaud et al. 2000). These effects are of particular concern to land managers as human populations near forests (wildland urban interface) are increasing (USDA 2004).

Attempts to ameliorate the effects of high-severity wildfires began as early as the 1920's, and were funded and implemented by federal and state land management agencies (Robichaud et al. 2000, Beyers 2004). The majority of these early attempts were focused on broadcast seed applications of native and non-native grasses and forbs with the intended goal of reducing erosion and increasing forage for domestic livestock (Robichaud et al. 2000, Beschta et al. 2003, Beyers 2004). In 1974, the federal government formalized a method for evaluating and implementing treatments designed to reduce the impacts of high-severity wildfires (Robichaud et al. 2000). This institutionalization of post-fire response resulted in the formation of interagency Burn Area Emergency Rehabilitation (BAER) teams to evaluate post-fire conditions. The stated purpose of these teams is to: "determine the need for and prescribe and implement emergency treatments to minimize threats to life or property or to stabilize and prevent further unacceptable degradation of natural and cultural resources resulting from the effects of a fire" (USDA 2004). Targeted areas for treatments are commonly high-severity portions of wildfires determined to have high potential for erosion, including

steep slopes that abut watersheds and tributaries deemed vulnerable to negative effects of post-fire erosion, and are determined to pose a threat to life or property (USDA 2004).

Treatments commonly applied to burned areas in an attempt to reduce runoff and erosion include mulching, felling of trees along slope contours, and broadcast seeding (Robichaud et al. 2000, USDA 2004). Seeding treatments are by far the most commonly and widely applied (Robichaud et al. 2000, Beschta et al. 2003, Beyers 2004, USDA 2004). Federal policy states that seed of native species should be utilized in post-fire seeding treatments whenever possible; however native seed is often unavailable, expensive, or originates from unknown genetic stock; therefore the inclusion of native seed in post-fire applications is limited (Richards et al. 1998, Robichaud et al. 2000, Beyers et al. 2004, USDA 2004). Presently, quick growing non-native annual cereal grains are most often seeded on wildfires (Richards et al. 1998, Robichaud et al. 2000, Beyers 2004, USDA 2004). The stated purpose of post-fire seeding is to establish vegetative cover as quickly as possible, with the goal to reduce bare ground and prevent or reduce erosion, to promote ecosystem rehabilitation, and to prevent invasion by non-native species (Robichaud et al. 2000, Beschta et al. 2003, USDA 2004).

Seeding treatments, bare ground, and vegetative cover

Seeding treatments are predicated upon the negative correlation between percent ground cover with runoff and erosion rates (Robichaud et al. 2000, Beyers 2004). Conceptually the relationship between cover and reduced erosion is twofold. First, standing crop and vegetative cover prevent rainsplash and sheet erosion by intercepting precipitation before it strikes the soil surface (Debano et al. 1998). Second, the roots of vegetation contribute to water infiltration into the soil and can aid in the deterioration of

hydrophobic soil layers that may result from high-severity wildfires, especially fires with long residence time as a result of high fuel accumulations (Debano et al. 1998). Several studies suggest that in a forested ecosystem with 75% of the ground covered by vegetation or litter, runoff and erosion are low, with only 2% of rainfall becoming surface runoff (Debano et al. 1998, Robichaud et al. 2000). In post-burn ecosystems with less than 10% ground cover, runoff can increase to over 70% with correspondingly high increases in erosion rates (Agee 1993, Debano et al. 1998, Robichaud et al. 2000). In many forested ecosystems, 60% ground cover appears to be sufficient to reduce erosion rates to approximate pre-fire conditions (Agee 1993, Robichaud et al. 2000). Thus, a target of 60% ground cover is often the primary response variable considered in evaluating the effectiveness of post-wildfire seeding (Robichaud et al. 2000, Beyers 2004).

Robichaud et al. (2000) used internal (Forest Service) reports and published articles to examine the effectiveness of post-fire seeding treatments. Vegetative cover data presented in the original 2000 report are summarized in Table 1, excluding all reports from non-forested ecosystems, and adding results from articles not published in the original report and those published since 2000. Of the 18 studies considered here (Table 1), three seeding treatments achieved 60% (+/-5%) vegetative cover, and one unseeded control achieved 60% (+/-5%) vegetative cover in the first year following fire. Three studies reported the difference between seeded and unseeded plots to be statistically significant, with seeded plots having higher total cover. In ten of the studies, the unseeded control had equal or higher total vegetative cover (+/-5%) compared to seeded treatments. The fact that less than 16% of reviewed studies achieved the target

60% vegetative cover calls into question the efficacy of this practice. In all of the studies (where information was provided) natural regeneration from ruderal species, seedbanks, and resprouting species was strong. This is illustrated by the 56% of the studies where unseeded areas had equal or higher vegetative cover compared to seeded areas.

Post-fire seeding studies in ponderosa pine forest are relatively few, and those that have been conducted show limited ability of seeded species to increase ground cover following fire. Of the studies considered here (Table 1), five are from ponderosa pine ecosystems; only one of these occurred in the semi-arid southwestern region of the United States (Arizona and New Mexico). Barclay et al. (2004) in New Mexico and Wagenbrenner et al. (2006) in Colorado ponderosa pine forest both examined the results of seeding treatments applied by the federal governments Burned Area Emergency Rehabilitation (BAER) program. Both of these studies examined above-ground vegetative cover and found no difference between seeded and unseeded areas in the first year following fire (Barclay et al. 2004, Wagenbrenner et al. 2006). Barclay et al. (2004) attributed low cover values to low precipitation over the study period.

United States federal policy mandates the inclusion of native seed into post-fire rehabilitation seed mixes when available and economically viable (Richards et al. 1998). One recent study found seeding with native species after wildfires was as effective as seeding with non-natives, although unseeded controls had comparable rates of vegetative cover in the first post-fire year (Thompson et al. 2006). This study was conducted after a high-severity wildfire with both native and non-native seed in pinyon-juniper (*Pinus edulis* Engelm.- *Juniperus osteosperma* [Torr.] Little) woodland and sagebrush steppe ecosystems in Utah. They found native seed produced higher or equivalent cover as non-

native seed mixes in the first and second year post-fire and that unseeded controls had equivalent or higher vegetative cover and density as areas seeded with native mixes (Thompson et al. 2006).

Other studies have examined the effectiveness of native seeding in ponderosa pine forest in the context of forest restoration including prescribed fire and forest thinning (Springer and Laughlin 2004, Daniels et al. 2008). These studies show increases in species richness as a result of seeding but no significant increase in vegetative cover as a result of seed additions (Springer and Laughlin 2004, Daniels et al. 2008), simply showing the species were able to establish but not demonstrating any functional advantage especially in the context of providing higher vegetative cover to prevent erosion. As in the case of Barclay et al. (2004), these studies attribute low vegetative cover following seeding to low precipitation (Springer and Laughlin 2004, Moore et al. 2006, Daniels et al. 2008), but other factors such as ungulate grazing may also influence post-fire vegetative cover (Sabo et al. 2009).

Ungulate grazing, like fire, is a form of disturbance that has differential impacts on species abundance and diversity (Oesterheld and Osvaldo 1990, Keeley et al. 2003). Livestock grazing has been linked with non-native species invasions and increased erosion (Fleischner 1994, Belsky et al. 1999). There is also a concern that livestock grazing will reduce establishment of post-fire vegetation and thus slow ecosystem rehabilitation (Beschta et al. 2003). Grazing can reduce growth of native species, especially at early growth stages (Gonzales and Arcese 2008). In northern Arizona, ponderosa pine forests ungulate livestock grazing has been shown to reduce richness and abundance of herbaceous plant communities (Bakker and Moore 2007). Grazing by elk

(*Cervus elaphus*) has also been shown to reduce growth of native vegetation following forest restoration treatments, such as forest thinning and prescribed burning (Huffman and Moore 2004).

Seed additions, plant community composition, and species richness

A stated purpose of post-fire seed additions is to promote ecosystem rehabilitation (Robichaud et al. 2000, USDA 2004). The concept that adding annual species to the post-fire environment will promote ecosystem rehabilitation is informed by the theory of directional succession which predicts post-disturbance landscapes will be dominated by ruderal species, followed by plants with different competitive and recruitment strategies (Allred and Clements 1949, Grime 1977). However, this may not be the realized pattern of post-fire plant regeneration, as mosaic burn patterns and residual plant communities may present different trajectories for plant community development (Platt et al. 2002). If mosaic burn patterns, residual plant communities, soil seed banks, and an influx of ruderal species all contribute to the post-fire plant community composition, then seed introductions may alter plant community development and have implications for community composition and potentially community trajectories in the short and long term (Schoennagel and Waller 1999, Keeley 2004).

Successional Models and Mosaic Burn Patterns

In its simplest form, the basic model of plant succession as proposed by Clements focused on directional changes in plant community composition following disturbance, or based on a theoretical ecological blank slate (Allred and Clements 1949). In this model, following a catastrophic disturbance such as a high-severity wildfire, species regeneration and compositional changes would proceed in a particular order and be determined by

specific plant life history traits based on reproductive and colonization strategies (Allred and Clements 1949, Grime 1977). Using Grime's (1977) terminology, ruderal (R-selected) species would be the first to colonize a disturbed area and would be followed by competitive (C-selection) species. Bradshaw (1984) used this Clementsian model of directional succession to predict plant community regeneration after severe disturbances. He predicted plant communities would follow a linear and parallel pattern of development from degraded to aggraded. In this model, ecosystem structure (species richness) would change in a parallel manner with ecosystem function (biomass and nutrient cycling) (Bradshaw 1984).

More recent research has shown that the complex dynamics of post-disturbance ecosystem regeneration and species recruitment and survival patterns do not fit well with Clements' linear succession model (Platt and Connell 2003, Anand and Desrochers 2004, Cortina et al. 2006). This may be especially true in post-wildfire ecosystems where seeding treatments are applied (Keeley 2004). Residual plant communities, differing effects on plant functional groups, and non-linear and negative relationships between species composition and function have been found (Bakker and Berendse 1999, Platt et al. 2002). High-severity wildfires generally result in non-catastrophic results; that is, with some patches of residual plant communities interspersed within the burn area, with non-uniform mosaic spatial patterns of disturbance (Platt et al. 2002). Mosaic disturbance patterns will result in islands of residual plant survivors available for colonization of burned areas and result in non-linear patterns of species recruitment and regeneration (Platt et al. 2002, Anand and Desrochers 2004).

Research in coniferous forests shows mixed results as to how seeding influences successional pathways and plant community development after fire. In a California ponderosa pine forest two years after wildfire and seeding, researchers found that cover and species richness of forbs, shrubs and conifers declined as the cover of the seeded grass (annual ryegrass, *Lolium multiflorum* Lam.) increased (Conard et al. 1991). This study showed that only two years after fire, plant species associated with later seral stages were already colonizing unseeded burned areas (Conard et al. 1991). Keeley (2004) showed seeding with wheat (*Triticum aestivum* L.) after a high-severity fire in ponderosa pine forest reduced species richness and cover of ruderal species most dramatically, but also reduced richness and cover of perennial species. These studies indicate seed additions may be co-opting the role of post-fire followers (Schoennagel and Waller 1999, Keeley 2004). In contrast, Kuenzi et al. (2008) examined plant regeneration in different burn severity classes after the large Rodeo-Chediski fire in northern Arizona and found that several years after fire seeded species had declined (Kuenzi et al. 2008). A primary difference between these studies is seeded species abundance, suggesting that when the seeded species produces enough vegetative cover to affect runoff and erosion rates, there may be detrimental effects to plant community development.

Post-fire Seed Introductions and Community Development

Annual cereal grains are often selected for post-fire seeding due to their qualities of quick growth, extensive fibrous root systems, and because they are non-reproducing or have low reproductive potential in forested ecosystems (Robichaud et al. 2000, USDA 2004). In ponderosa pine forests, post-fire seeded species can have temporary high abundance values and then decline over time (Schoennagel and Waller 1999). However,

there is debate as to whether these seeded species inhibit or facilitate plant community development (Beyers 2004, Barclay et al. 2004, Keeley 2004).

There is some evidence that seeding may reduce the abundance of fire-following species. Ruderal forbs are an important part of the post-fire plant community in many forested ecosystems (Laughlin et al. 2004, Keeley 2004), and may be reduced in abundance in response to post-fire seeding (Barclay et al. 2004, Schoennagel and Waller 1999). A reduction in the abundance of ruderal species may be desirable if the colonizers are non-native invaders (Thompson et al. 2006); in contrast, such reductions may be deleterious to local biological diversity if ruderal fire-followers are natives or endemics whose life cycle depends on the post-fire environment (Keeley 2004).

Competition from seeded species can reduce abundance of long-lived foundation species. In many plant communities, dominant conifers and shrubs are considered foundation species and as such have a large influence over many ecosystem processes, community composition, and may stabilize abiotic components of ecosystems (Buonopane et al. 2005, Ellison et al. 2005). Thus reestablishing these dominant plants within an ecosystem is important for restoring community composition, structure, and long-term ecosystem rehabilitation. However, several studies documented decreased conifer establishment in seeded plots (Schoennagel and Waller 1999, Kruse et al. 2004, Keeley 2004). These studies hypothesized that seeded species out-competed conifers for resources (Schoennagel and Waller 1999, Anderson and Brooks 1975). Amaranthus et al. (1990) suggested that depression of mycorrhizal associations may be a contributing factor of decreased shrub and conifer establishment in post-fire seeded areas. Seeded annual grasses do not support ectomycorrhizal fungi important to conifers and may have

deleterious effects on pine seedling establishment; these relationships vary more for shrubs and effects will depend on the shrub species present in the plant community (Amaranthus and Perry 1987, Amaranthus et al. 1990). Together, high dominance and out-competing of shrubs and conifers for resources, coupled with reductions in mycorrhizal associations, may be sufficient to explain reductions in shrub and conifer abundance in some post-fire seeding studies.

Seeding quick-growing annual grasses after high-severity fire relies on the concept that these plants will stabilize the ecosystem, reduce erosion, and facilitate plant community regeneration (USDA 2004). The idea that non-native seeding promotes ecosystem rehabilitation has been contradicted by several post-fire and post-disturbance studies. Zelnik et al. (2008) conducted roadside revegetation experiments with seed mixes that were comprised mostly of native species but containing varying percentages of annual cereal grains. They found plots seeded with annual non-natives delayed plant community regeneration and reduced species richness (Zelnik et al. 2008). Reduced species richness as a result of seeding has been documented in ponderosa pine (Anderson and Brooks 1975, Barclay et al. 2004, Keeley 2004), pinyon-juniper woodlands (Thompson et al. 2006), and chaparral ecosystems (Taskey et al. 1989, Keeley et al. 2006). Several studies demonstrate seeded species dominance may reduce species richness (Conard et al. 1991, Barclay et al. 2004, Hunter and Omi 2006). A mechanism explaining species richness reduction as related to biomass was presented by Grime (1973) and again by Grace (1999). They both examined herbaceous plant communities and hypothesized that habitats with very high productivity would have low species densities. In a post-fire seeding study, Keeley (2004) observed a reduction in species

richness from an average of 152 species in unseeded plots to an average of 104 in plots seeded with wheat, thus offering support for Grime's (1973) and Grace's (1999) hypothesis. He also reported an average percent vegetative cover of 95% in seeded plots, of which 67% was contributed by the seeded wheat (Keeley 2004), indicating seeded species dominance likely plays an important role. Tilman (1995) further discussed competition theory and argued that a primary force structuring plant communities is competition between species for resources. Hunter and Omi (2006) supported this idea; they correlated high post-fire dominance and productivity of seeded species with increases in light and nutrient availability. This suggests quick-growing seeded annuals have a competitive advantage in post-fire landscapes (Hunter and Omi 2006).

Seeding and non-native species abundance and richness

Post-wildfire seeding treatments are frequently designed to reduce or prevent non-native species invasions (Robichaud et al. 2000, USDA 2004). High-severity wildfires consume aboveground biomass and increase the availability of light and nutrients (Ryan and Noste 1985, Debano et al. 1998). As a result of this nutrient-rich post-fire environment, increases in abundance and richness of ruderal native and non-native plants is well documented in many forested ecosystems, including southwestern ponderosa pine forests (Crawford et al. 2001, Keeley et al. 2003, Keeley 2004, Dodge et al. 2008, Laughlin and Fulé 2008).

The concept that seeded species will reduce non-native species is based on the notion that these quick growing, short-lived, seeded annuals will utilize a high percentage of available resources thus co-opting them from invading non-native species (Grace 1999, Robichaud et al. 2000, Grime 2001, Beyers 2004). Keeley (2004) compared

seeded and unseeded areas and noted a reduction in non-native species richness and cover in the first year following fire in a California ponderosa pine forest seeded with winter wheat. Barclay et al. (2004) studied a high-severity burn in a ponderosa pine forest, comparing areas seeded with ryegrass to unseeded areas, and found native plant cover, total species richness, and cover of non-natives to be lowest in seeded plots. There is concern that the disappearance of short-lived seeded annuals in the second or third post-treatment growing season may leave open ecological niches that are vulnerable to invasion by non-native species in following years (Mooney and Hobbs 2000, Keeley 2004). To address this, a few seeding studies in ponderosa pine forest have experimented with native perennial seeding in conjunction with forest restoration (thinning and burning). These studies found seeding did little to affect non-native species abundance or non-native richness (Springer and Laughlin 2004, Daniels et al. 2008).

Two distinct patterns of non-native species recruitment following fire have been found from post-fire research in ponderosa pine forests. The first indicates non-native species increase following fire and these increases may be positively correlated with fire severity (Crawford et al. 2001, Griffis et al. 2001, Barclay et al. 2004, Hunter and Omi 2006). The second demonstrates that while ruderal species increase following fire there is not a significant increase in non-native species (Huisinga et al. 2005, Kuenzi et al. 2008). A significant portion of the differences in conclusions between these studies is attributable to the floristic treatments of the species they documented. Species that occupied a large portion of the post-fire plant community in both Crawford et al. (2001) and Barclay et al. (2004) are common regionally and classified as native in other research and from other floristic sources (Huisinga et al. 2005, USDA Plants 2006, Kuenzi et al.

2008). These inconsistencies indicate that non-native invasions may not be the inevitable result of high-severity fires and that the role of seeding in decreasing invasions is unclear.

Impacts of Seeding Native Species on Local Gene Pools

Local genetic adaptations may occur in many plants and can be a source of variation that allow local populations to be adapted to site specific conditions (Grant 1982). Raabova et al. (2007) demonstrated local adaptation at small scales. They used several common gardens and showed seed germination success was highly correlated to site-specific conditions where seed germinated best in its place of origin (Raabova et al. 2007). Other experiments have shown this pattern of local adaptation in species that occur over broad regional landscapes (Millar and Libby 1989). Several species have within-population variations at the regional scale that infers differential success in outplanting experiments (Millar and Libby 1989). The risk of seeding with native species that are not locally adapted is twofold. First, the seed may be poorly adapted to site-specific conditions and germination and success of seeding may be reduced (Raabova et al. 2007). Second, gene flow from introduced plants may swamp local populations, effectively removing or altering patterns of local adaptations (Jones 1998, McKay et al. 2005).

Research investigating the efficacy of seeding with native species following high-severity wildfires is generally lacking (Beyers 2004). Several studies have reported on seeding of native hybrids following fire (Barclay et al. 2004, Hunter and Omi 2006). Hybrids are selected for specific growth attributes and may not be a valid representation of how well a particular native species would grow if utilized in post-wildfire seeding applications. Seeding with non-local sources will have the least effect on native gene

pools and the highest likelihood of success if species selected have low genetic variability over the entire species range (Millar and Libby 1989).

Future Research

The role of seeding after high-severity wildfires in northern Arizona is generally understudied. Of the published studies considered here, only four were conducted in ponderosa pine forest (Conard et al. 1991, Barclay et al. 2004, Keeley 2004, Wagenbrenner et al. 2006). Two of those were in southwestern portion of the ponderosa pine range: one in Colorado (Wagenbrenner et al. 2006) and one in New Mexico (Barclay et al. 2004). With one exception (Thompson et al. 2006), all of the post-fire seeding studies considered here were observational, in that they simply established study plots where BAER teams applied seeding treatments. BAER seeding is generally via aerial applications and thus seed spread may be uneven over the landscape. Further, unseeded control plots with similar burn severity, aspect, elevation, and forest types may be difficult to establish. There is very little understanding of the effectiveness of native seed mixes in the context of post-fire applications. Further research is required to determine the efficacy and effectiveness of post-wildfire seeding, examine the relationship between post-fire seeding and non-native species, and test if seeding alters post-fire plant community composition.

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Table 1.Total plant cover on seeded and unseeded treatments and plant cover of seeded species in the first year following high-severity wildfires from 18 unpublished and published papers, by location and plant community type. The first nine were originally reported in Robichaud et al. (2000) and last nine studies were either not published in the original review or published since the original review. Cover values are treatment averages if given, otherwise ranges (low – high) are shown. Statistical difference between treatments, when reported is signified by (*).

Study Location	Vegetation Type	Treatment Type	Seeded SpeciesCover / Total Cover	Unseeded Control % Cover	Source
Siskiyou Mts. Or.	Douglas-fir	annual ryegrass	49 / 50*	9	Amaranthus 1989
Siskiyou Mts. Or.	Douglas-fir	annual ryegrass	85.2 / 87.1*	23.6	Amaranthus 1989
near Loman, Id.	Douglas-fir	non-native mix[a]	1 / 14	15	Geier-Hayes 1997
near Loman, Id.	ponderosa pine	non-native mix[a]	3 / 10	nd	Geier-Hayes 1997
near Loman, Id.	subalpine fir	non-native mix[a]	nd / 7	nd	Geier-Hayes 1997
near Greenville, Ca.	mixed conifer	non-native mix[b]	nd / 6	7	Roby 1989
Entiat Exp. Forest, Wa.	ponderosa pine - Douglas-fir	non-native mix[c]	3.3 / 10.3	5.6	Tiedemann and Klock 1973
Snow Basin, Or.	pine-mixed fir	non-native mix[d]	12 / 44*	12	Anderson and Brooks 1975
Santa Lucia Mts., Ca.	sugar pine	annual ryegrass	5 to 70 / 10 to 75	5	Griffin 1982
Greenville, Ca.	mixed conifer	non-native mix[a]	10 / 24	27	Ruby 1989
Pocatello, Id.	sagebrush-juniper-grassland	non-native mix[b]	nd / 12.8	18.3	Ratzlaff and Anderson 1995
Eastern Cascades, Wa.	grand fir (Abies grandis)	non-native mix	nd / 41.5	48	Schoennagel and Waller 1999
N.Western Ca. (Megram Fire)	mixed-conifer	barely + mulch	nd / 31.1	24.3	Kruse et al. 2004
Jemez Mtns., NM.	ponderosa pine	annual ryegrass	nd / 16.7	37.5	Barclay et al. 2004
Central Sierras, Ca. (highway fire)	mixed ponderosa pine -oak-	common wheat	67 / 95*	55	Keeley 2004
N.Central Wa. (North 25 Fire)	mixed-conifer	common wheat	4.5 / 18	18	Robichaud et al. 2006
Tintic Valley, Ut.	pinyon juniper	non-native mix[c]	4 / 4.3	7	Thompson et al. 2006
Loveland, Co.	ponderosa pine	non-native mix[e]	nd / 28	43	Wagenbrenner et al. 2006

Seed Mixes: [a]Orchardgrass, tall fescue, timothy, slender wheatgrass, [b]intermediate wheatgrass, Siberian wheatgrass, medacago sativa, [c]crested wheatgrass, Russian wildrye, tall wheatgrass, western wheatgrass, [d]Great Basin wildrye, bluebunch wheatgrass, Indian ricegrass, squirreltail, western wheatgrass, [e]annual cereal grains

CHAPTER THREE

Minimal effectiveness of native and non-native seeding following three high-severity wildfires

Abstract

The rationale for seeding following high-severity wildfires is to enhance plant cover and reduce bare ground, thus decreasing the potential for soil erosion and non-native plant invasion. However, experimental tests which measure seedings' effectiveness in meeting these objectives in forests are lacking. We conducted three experimental studies of the effectiveness of seeding with non-native (annual ryegrass, *Lolium multiflorum* (Lam.) and wheat, *Triticum aestivum* L.) and native species (squirreltail (*Elymus elymoides* (Raf.) Swezey), blue grama, (*Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths), muttongrass (*Poa fendleriana*, (Steud.) Vasey), scarlet gilia (*Ipomopsis aggregata* (Pursh) V.E. Grant), and purple locoweed (*Oxytropis lambertii*, Pursh.)) on three Arizona wildfires. Seeding treatments were largely ineffective in increasing vegetative cover or decreasing exposed bare ground. At one treatment at one fire, wheat seeding at the Warm Fire, senesced seeded annuals increased litter cover and resulted in lower bare-ground values than unseeded controls. Only on one fire did seeded non-native annuals establish well, resulting in 20-29% vegetative cover. On the other two fires, seeded cereal grains accounted for <3% cover. At all fires, native seeded species contributed between <1 and ~12% vegetative cover. On the Warm Fire, total plant cover was higher on both areas seeded with annual ryegrass (53% cover) and those seeded with a native seed mix (49% cover), as compared to unseeded controls (41% cover). Vegetative cover on all treatments, including unseeded treatments, was at or near 40% the first year

following fire, at all three study sites. Non-native species richness and abundance did not differ among treatments at any fire. We attributed the ineffectiveness of seeding in altering the abundance or richness of non-native species in the first year to two factors. First, non-native abundance and richness was low on all three fires. Second, even on the most successful seeding treatments on the Warm Fire, resulting plant cover was inadequate to exclude non-natives. This study adds to growing evidence that post-fire seeding is ineffective in enhancing post-fire plant cover and reducing invasive non-native plants.

Keywords: Burned Area Emergency Rehabilitation, Non-native plants, ponderosa pine, wildfire

Introduction

High-severity wildfires in forested ecosystems consume high percentages of above-ground biomass, vegetative cover, litter, and duff (Ryan and Noste 1985). The increase in bare mineral soil, light, and nutrient availability following high-severity wildfires may increase water runoff, soil erosion, and facilitate non-native plant invasions (Benavides-Solorio and MacDonald 2001, Crawford et al. 2001, Johansen et al. 2001). In an attempt to ameliorate the effects of high-severity wildfires, resource managers frequently apply post-wildfire treatments, such as contour felling of logs, mulching, and seeding. Due to its low cost, ease of application, and long history of use, seeding is the most widely applied, post-wildfire treatment (Beyers 2004). Species used in post-wildfire seeding are most commonly quick-growing non-native grasses (Robichaud et al. 2000). Quick growth is important because the risk of increased runoff and erosion is highest in the first year post-fire (Robichaud et al. 2006, Wagenbrenner et al. 2006). Seeded species most commonly utilized include non-native cereal grains, which are either sterile hybrids or annuals with low reproductive potential in forested ecosystems (Robichaud et al. 2000). Seeding with longer-lived native perennial species has been recommended as a method for producing more persistent vegetative cover and standing crop compared to seeding with annuals (Richards et al. 1998, Beschta et al. 2003). However, use of native seeded species in post-wildfire rehabilitation has been limited and there is relatively little published information on the effectiveness of native species in meeting post-fire vegetative cover objectives (Bescheta et al. 2003).

Post-wildfire seeding is predicated on the assumption that seeding will enhance plant cover and reduce bare ground, thereby reducing runoff and erosion rates

(Robichaud et al. 2000, Beyers 2004). Conceptually the relationship between cover and reduced erosion is twofold. First, plant cover prevents rainsplash and sheet erosion by intercepting precipitation before it strikes the soil surface (Debano et al. 1998). Second, plant roots increase soil water infiltration and can aid in the deterioration of hydrophobic soil layers that may result from high-severity wildfires (Benavides-Solorio and MacDonald 2001).

The proportion of exposed bare ground and ground covered are important predictors of post-fire runoff and erosion rates in coniferous forests. Several studies from forested ecosystems have demonstrated significant positive correlations between the percentage of exposed bare ground and both water runoff and soil erosion (Johansen et al. 2001, Benavides-Solorio and MacDonald 2001). These studies have shown consistent significant increases in soil loss as exposed soil reaches > 60-70% on severely burned areas (Johansen et al. 2001). Other research has shown a negative correlation between ground cover as compared to post-fire water runoff and soil erosion (Johansen et al. 2001, Wagenbrenner et al. 2006). Post-fire erosion studies from coniferous forests support the concept that high cover values from plant regeneration result in low amounts of runoff and erosion rates (Johansen et al. 2001, Wagenbrenner et al. 2006). Recent studies from coniferous forests including, Colorado ponderosa pine forests and from grand fir (*Abies grandis* Douglas ex D. Don) forests, have demonstrated the range of cover most closely associated with low amounts of runoff and erosion is between 50 -70% (Johansen et al. 2001, Robichaud et al. 2006, Wagenbrenner et al. 2006). Researchers and land managers commonly use 60% cover as a target value in evaluating the effectiveness of seeding treatments (Robichaud et al. 2000, Beyers 2004, Keeley 2004, USDA 2004a).

Preventing or reducing the abundance of invasive non-native species after fire is an important management goal of many seeding efforts (Robichaud et al. 2000, Beyers 2004). This is based on the idea that seeded species will grow quickly and utilize available resources, thus co-opting them from invading non-natives. Previous research has demonstrated the degree to which the seeded species are dominant in the plant community may be an important predictor of how effectively non-native species abundance is reduced. Several studies from both wildfires and prescribed burning experiments have shown seeding, whether with native or non-native species, had little effect on non-native species abundance (Kruse et al. 2004, Springer and Laughlin 2004, Daniels et al. 2008, Kuenzi et al. 2008). What these studies have in common is that they reported relatively low dominance of seeded species in the plant community. In contrast, two recent studies demonstrated at least short-term reductions in non-native species following post-wildfire seeding with either wheat (Keeley 2004) or native seed mixes Thompson et al. 2006); in both cases, seeded species had high dominance in the post-fire plant community.

Experimental tests of the effectiveness of post-wildfire seeding of native and non-native species in enhancing plant cover and standing crop, reducing bare ground, and curtailing invasive non-native plants are rare. Thompson et al. (2006), who experimented with native and non-native seeding in a sagebrush steppe plant community, is a notable exception, but we were unable to find similar studies from coniferous forests. Most studies have been opportunistic and observational, in that researchers have established study areas within large landscape level seeding treatments implemented by managers following fire. This has led to difficulting in locating unseeded control areas for

comparisons as well as uncertaintiy over the amount of seed applied at the plot level of observation (Schoennagel and Waller 1999, Barclay et al. 2004, Kruse et al. 2004). The result of these difficulties can be small sample sizes and low certainty of conclusions.

For this study we took advantage of three separate, high-severity wildfires which occurred in ponderosa pine forests. In order to determine the effects and effectiveness of native and non-native seeding in the first year following fire we used a controlled, replicated, and randomized experimental design. We hypothesized that seeding would: 1) decrease bare ground; 2) increase plant canopy cover and standing crop; and 3) reduce non-native species abundance and richness.

Methods

Study areas

We selected three wildfires in Arizona ponderosa pine (*Pinus ponderosa* C. Lawson) forests for this study (Figure 1). The Potato and Warm Fires burned in 2006; the Birdie Fire burned in 2007. We sampled immediately post-burn and for a full year after fire, the period within which plant establishment is required to meet rehabilitation objectives. Replicated, randomized, controlled experimental units were established in areas dominated by ponderosa pine, within four weeks of fire containment, and were located within high-severity portions of the burn, as defined by federal Burned Area Emergency Rehabilitation (BAER) teams. Severity classification was ground-truthed by visually assessing tree mortality, crown scorch height, and ground char (Ryan and Noste 1985).

The fires spanned an elevational and latitudinal gradient (Figure 1). The Warm Fire burned from 8 June to 4 July 2006 on the Kaibab National Forest. This fire had the

highest average elevation, 2400 m; the greatest average yearly precipitation, 64.5 cm; and occurred on Mollic Eutroboralf soils derived from a limestone parent material (Brewer et al. 1991, WRCC 2009). The Potato Fire burned from 6-28 June 2006 on the Apache-Sitgreaves National Forest. This fire had the lowest average elevation, 2050 m; the lowest average yearly precipitation, 52.4 cm; and occurred on Mollic Eutroboralf soils derived from basaltic parent material (Laing et al. 1991, WRCC 2009). The Birdie Fire burned from 6-15 July 2007 on the Coconino National Forest. The average elevation, 2200 m, and average annual precipitation, 59 cm, are intermediate to those at the Warm and Potato Fires (Miller et al. 1991, WRCC 2009). Soils at the Birdie Fire are classified as Mollic Eutroboralfs from basaltic parent materials (Miller et al. 1991). Precipitation at all three sites approximated long term averages in the year of fire and the first year following fire (Figure 2). Fire containment, plot establishment, and seeding all occurred within one to three weeks of the onset of high intensity summer monsoonal precipitation (Figure 2, WRCC 2009).

Experimental Design and Treatments

We established fifteen 20- X 25-m (500-m^2) plots per treatment at each fire. Plots were located within unseeded areas of high-severity burn, with slopes between 5 and 25%. The overstory was dominated by ponderosa pine prior to the fires. Data were collected immediately following fire and in the first of the following year. Experimental treatments were: 1) seeding with common wheat (*Triticum aestivum* L.); 2) seeding with annual ryegrass (*Lolium multiflorum* Lam.); 3) seeding with a native seed mix including: squirreltail (*Elymus elymoides* (Raf.) Swezey), blue grama, (*Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths), muttongrass (*Poa fendleriana*, (Steud.) Vasey), scarlet gilia

(*Ipomopsis aggregata* (Pursh) V.E. Grant), and purple locoweed (*Oxytropis lambertii*, Pursh.); plus 4) unseeded control. Due to fire size treatments varied between fires. At the Potato and Birdie Fires, treatments 1, 3, and 4 were tested; at the Warm Fire all four treatments were tested. Our target seeding rate was 403 pure live seeds (PLS)/m^2 (Table 1), based on the USDA (2004b) recommended rate of 269 PLS/m^2 with an increase of 1.5 to 2 times for broadcast seeding. These seeding rates are similar to those used in other post-fire seeding experiments (Robichaud et al. 2000, Thompson et al. 2006). Seed was hand broadcast at each plot immediately following plot establishment.

Species used in seed treatments were selected because they are commonly used by land managers or show promising results in studies in this region (Jones 1998, Robichaud et al. 2000, Elseroad et al. 2003, Barclay et al. 2004). Annual ryegrass was seeded on portions of the Warm Fire and the 1997 Dome Fire in New Mexico (Barclay et al. 2004). Wheat was seeded on portions of the Potato Fire and the 2002 Rodeo-Chediski Fire in Arizona (Kuenzi et al. 2008). We included species with broad geographic distributions and included both cool-season (C3: bottlebrush squirreltail,muttongrass, and purple locoweed) and warm-season (C4: blue grama and scarlet gilia) plants in an effort to enhance plant cover throughout the growing season (Dodge 2004, Springer and Laughlin 2004, Moore et al. 2006, Daniels et al. 2008).

Response Variables

We estimated total plant canopy cover, litter, rock, and bare ground cover, plus cover by life form (graminoids, shrubs, and forbs) and by species in fifty, 20- X 50-cm (0.1-m^2) quadrats per plot. We placed the quadrats at 1-m intervals on the right side of 5 permanent transects in each plot. We estimated cover by classes (1 = <1%, 2 = 1-5%, 3 =

6-25%, 4 = 26-50%, 5 = 51-75%, 6 = 76-95%, 7 = 96-100%) modified from Daubenmire (1959). We added the <1% class based on Bailey and Poulton's (1968) recommendation to better describe vegetation occurring in very small amounts (e.g., 0-1 percent). We used mid-points of each cover class to calculate averages.

Plant standing crop was sampled in each plot by clipping herbaceous vegetation in fifteen, 0.25-m^2 circular quadrats, three per transect, randomly placed along the left side of each transect. Clipped material was separated by species in the field, placed in paper bags, oven dried for 48 hours at 70˚ C, and weighed to the nearest 0.01 g (Moore et al. 2006).

All sampling was conducted between the end of July through the end of August, in order to capture plant cover and biomass present at the onset of high intensity monsoonal rainstorms (WRCC 2009). These rainstorms account for approximately half of yearly precipitation in this region (WRCC 2009). Plant nomenclature and nativity are according to the Plants Database (USDA NRCS 2009). Plant reference specimens are stored at the Ecological Restoration Institute at Northern Arizona University, Flagstaff, Arizona.

Statistical Analysis

Due to differences in site characteristics, dates of burn, and variations in treatments applied, we analyzed each fire as a separate experiment. We tested assumptions of normal distribution of data using Shapiro-Wilk goodness of fit test and homogeneous variances using Levene test in JMP V 7.0.1 (SAS institute Inc. 2009). Due to non-normality and heterogeneous variances in many of our response variables we used nonparametric testing procedures (Anderson 2001).

We used PERMANOVA analysis in PC-ORD V.5.1 with Bray-Curtis distance measures for all analyses, although similar results were observed using Euclidean distances (Ludwig and Reynolds 1988, Faith et al. 1987, McCune and Mefford 1999). PERMANOVA is a distance-based nonparametric analysis of variance procedure that can be used with univariate or multivariate datasets. We used 9999 permutations per test (Anderson 2001). When significant differences were detected among treatments, we used a nonparametric pair-wise *a posteriori* t-test to separate means (Anderson 2001). For all tests we used $\alpha = 0.05$ to determine significance.

We tested for differences among treatments immediately post-fire and in the first year following fire. Immediately post-fire, but before seeding treatments were applied, we tested for differences in substrate, litter, and plant cover among treatments. In the first year following fire, we tested for differences among treatments in total plant cover, standing crop and cover of non-native species (excluding seeded non-natives), litter, and bare ground. We also analyzed cover and standing crop by life-form (graminoids, forbs, shrubs) by treatments.

Results

Does seeding decrease bare ground?

Immediately post-fire, before treatments were applied, bare ground ranged from 42-48% at the Warm Fire, 65-68% at the Potato Fire, and 55-60% at the Birdie Fire (all SE's <5.2%). There were no significant differences in bare ground among treatments at the Potato, Birdie, or Warm Fires (P = 0.86, 0.89, 0.76 respectively). Litter, rocks, and logs accounted for the remainder of ground cover. Vegetative cover averaged between <0.5- 5.0%, with no significant difference among treatments at any fire.

We found only partial support at one fire for the hypothesis that seeding significantly decreased bare ground. In the first year after seeding, bare ground did not differ among treatments at the Potato (P = 0.99) or Birdie Fires (P = 0.77). Bare ground differed among treatments only at the Warm Fire, with significantly (P = 0.03) lower levels on plots seeded with wheat than unseeded treatments (Figure 3A). This difference was explained by higher levels of litter at the Warm Fire wheat treatments as compared to the unseeded treatments (P = 0.02). Litter cover did not differ among treatments at the Potato (P = 0.91) or Birdie Fires (P = 0.88).

Does seeding increase ground cover, plant cover, and standing crop?

Warm Fire

Results from the Warm Fire in the first year following fire offered partial support for the hypothesis that seeding increases plant canopy cover but did not support the hypothesis that seeding increases plant standing crop. Total canopy cover differed by ~13% among treatments and was significantly higher on ryegrass seeded plots (P <0.01) and native seeded plots (P = 0.04) compared to unseeded plots (Figure 3B). However, total plant standing crop did not differ among treatments (P = 0.28) (Figure 4A).

Seeding altered the abundance of graminoids. Graminoid cover and standing crop differed significantly (P = 0.02 and <0.01) among treatments, with higher graminoid cover and standing crop on seeded plots compared to unseeded controls. Annual ryegrass and wheat were major species in terms of canopy cover on their respective treatments (Table 2). On the native seed treatment, two seeded grasses, squirreltail and muttongrass were dominant in terms of plant cover and standing crop. Major species in terms of plant canopy cover and standing crop on unseeded plots included squirreltail, muttongrass, a

native *Carex*, and a variety of native forbs. Graminoid standing crop on the unseeded controls was the lowest at 54.4 kg ha^{-1} (SE 12.5) compared to the native seeded treatment at 114.2 kg ha^{-1} (SE 21.2), the ryegrass at 163 kg ha^{-1} (SE 36.1), and the wheat treatment had the highest at 368.8 kg ha^{-1} (SE 97.2). Total forb cover, standing crop, and shrub cover did not differ significantly among treatments (P = 0.51, 0.29 and 0.12) (Figure 4A).

Potato Fire

Data from the Potato Fire did not support the hypothesis that seeding after high-severity wildfires increases vegetative cover or standing crop. Total plant canopy cover (P = 0.43) and plant standing crop (P = 0.33) did not differ significantly among treatments in the first year following fire (Figure 3B, 4B). Similarly, we did not detect significant differences in canopy cover (P = 0.33) or standing crop (P = 0.48) of graminoids among treatments (Figure 4B). Nor was there a significant difference in total forb cover (P = 0.51) or standing crop (P = 0.36) among treatments. There were only trace amounts of shrub species.

In contrast to the Warm Fire, species included in the seed mixes were not always the major species in terms of plant canopy cover. Wheat cover averaged <3% on the wheat seed treatment, and on the native seed treatment, only two seeded species, blue grama and squirreltail, had cover >2% (Table 2B). Instead, native forbs originating from natural regeneration and recruitment dominated plant canopy cover on all treatments the first year following burning.

Birdie Fire

Data from the Birdie Fire did not support the hypothesis that seeding after high-severity wildfires increases vegetative cover or standing crop. Total vegetative cover (P

= 0.31) and standing crop (P = 0.74) did not differ among treatments in the first year following fire (Figure 3C). However, graminoid cover and standing crop differed among treatments (P = 0.03 and 0.01), with higher levels on native seeded treatments compared to both unseeded and wheat treatments. Graminoid standing crop on the unseeded controls was 58.5kg ha^{-1} (SE 15.9) compared to 91.8 kg ha^{-1} (SE 41.2) on the native seeded treatment, and 59.5 kg ha^{-1} (SE 18.7) on the wheat treatment (Figure 4).

The Birdie Fire was similar to the Potato Fire in that wheat had very low cover in the first year following fire (~2%) (Table 2C). On the native seeding treatment, two seeded species, blue grama and squirreltail were major species. On unseeded plots, native forbs and graminoids constituted the majority of the post-fire plant cover.

Does seeding reduce non-native species?

Contrary to our hypothesis, seeding did not reduce the abundance or richness of non-native species the first year post-burn. Plant standing crop of non-native species was highly variable and did not differ among treatments at any fire (Potato fire: P = 0.17, Birdie Fire: P = 0.82; we detected only trace amounts of non-natives at the Warm Fire). Non-native species richness (other than the seeded species) ranged from 4 to 11 on the three fires and did not differ among treatments on any fire.

In the first year post-burn at the Warm Fire, we found five non-native species: cheatgrass (*Bromus tectorum* L.), bull thistle (*Cirsium vulgare* (Savi) Ten.), redstem stork's bill (*Erodium cicutarium* (L.) L'Her. Ex Aiton), common dandelion (*Taraxacum officinale* F.H. Wigg.), and yellow salsify (*Tragopogon dubius* Scop.). Richness of non-natives among treatments was similar, with the exception of yellow salsify, which was not present on the native seeded treatments.

Non-native species at the Potato Fire in the first year following fire were bull thistle, redstem stork's bill, little hogweed (*Portulaca oleracea* L.), and common mullein. Little hogweed was prevalent on all plots but due to its diminutive stature, accounted for little standing crop. Bull thistle was absent from the native seeded plots. All these species made up trace (<2% of total non-native standing crop) amounts except mullein, which was responsible for the majority of the non-native standing crop. Distribution and abundance of mullein varied among treatments but not significantly due to the large variation. Mullein occurred on one third of the total plots and accounted for an average of between 7-30% of total standing crop (mullein standing crop: unseeded plots (12.7 kg ha^{-1} SE 8.5), native (40.6 kg ha^{-1} SE 30.8), and wheat plots (102 kg ha^{-1} SE 68).

Eleven non-native species were present at the Birdie Fire (cheatgrass, bull thistle, prickly lettuce, prostrate knotweed (*Polygonum aviculare* L.), little hogweed, Canada bluegrass (*Poa compressa* L.), Kentucky bluegrass (*Poa pratensis* L.), Dalmatian toadflax (*Linaria dalmatica* L. Mill.), common dandelion, yellow salsify, and common mullein). Two species occurred only on native seeded plots (yellow salsify and Canada bluegrass). The two most abundant non-natives were Kentucky bluegrass and common mullein; together they accounted for over 90% of non-native standing crop at all treatments. Similar to the Potato Fire, the distribution of these species varied greatly, occurring on less than one third of the plot. When these species occurred, they accounted for between 21- 40% of total standing crop.

Discussion

Does seeding decrease bare ground?

Immediately post-fire, the percentage of bare ground on two fires was above 55%, thus potentially putting these sites at risk for increased sediment loss. On the Warm Fire, percentage of bare ground was lower, but still within the range considered to be at risk for increased sediment yields following especially intense storms (Robichaud et al. 2000, Wagenbrenner et al. 2006). A year later, the percentage of bare ground on all treatments at the Potato Fire changed very little and remained at levels considered to be at risk for increased erosion. The percentage of bare ground at the Birdie Fire, regardless of treatment, had declined to levels considered to be low enough to curtail erosion (Johansen et al. 2001). At the Warm Fire, wheat seeding significantly decreased bare ground compared to all other treatments, largely due to increased litter cover from the senesced wheat. Levels of bare ground at the Warm Fire on all treatments, including unseeded controls, were low enough to protect sites from all but the most intense storms (Johansen et al. 2001, Pannkuk and Robichaud 2003, Groen and Woods 2008). Thus, with the exception of wheat seeding on one fire, seeding treatments were largely ineffective in decreasing bare ground. At two fires, litter accumulations and plant regeneration on all treatments, including unseeded treatments, resulted in exposed bare ground values well below those associated with increased runoff and erosion (Johansen et al. 2001).

The ineffectiveness of seeding treatments at reducing bare ground compared to the natural regeneration that we documented is similar to several previous post-wildfire studies (Robichaud et al. 2006, Wagenbrenner et al. 2006). In fact, evidence supporting the effectiveness of seeding in significantly reducing bare ground is limited. Amaranthus

et al. (1993) reported significant reductions in bare ground following seeding with annual ryegrass, and Keeley (2004) reported similar results with wheat seeding at high rates. In both of these studies, and in our one treatment where bare ground was lower, the resulting reductions in bare ground were attributable to a thatch layer produced by the senesced seeded annuals.

Does seeding increase plant cover and standing crop?

Our results, along with previous research, indicate the ability of post-wildfire seeding treatments to significantly increase vegetative cover in forested ecosystems in the first year following fire is minimally supported. Less than one quarter of the post-fire seeding studies from forested ecosystems considered in Robichaud et al.'s (2000) review, as well as those published since 2000, report significant increases in vegetative cover as a result of seeding (Table 3). Where we did find differences in cover resulting from seeding, the unseeded treatments averaged 40% cover, whereas the native treatments averaged 49%, and the ryegrass averaged 53% cover. However, whether these differences in vegetative cover will confer substantial reductions in post-fire runoff and erosion is unclear (Johansen et al. 2001, Benavides-Solorio and MacDonald 2001). Using the target cover value of 60%, no treatment achieved this level of cover in the first year following fire.

Although seeding is predicated on the assumption that plant regeneration following high-severity fire will be slow and seeding is necessary to increase cover we found relatively high amounts of vegetative cover on all of the unseeded treatments at all three case studies (Robichaud et al. 2000). Plant species utilizing multiple regeneration and recruitment strategies were present in the post-fire plant community. These included

perennial plants regenerating from underground persistent root masses, regeneration of species that are known to produce persistent seed banks, and colonizing species (Wienk et al. 2004, Korb et al. 2005, USDA 2009).

Standing crop sampling accounts for differences in plant architecture that were not captured in vegetative cover values; for example annual ryegrass is relatively prostrate (high relative cover and low standing crop) whereas wheat has a vertical architecture (low relative cover and high standing crop). At one fire the increased graminoid standing crop we recorded was attributable to seeded annual species. Seeded annuals complete their lifecycle in one growing season, senesce, and in following growing seasons can contribute to increased litter but not to belowground biomass (Amaranthus et al. 1993, Schoennagel and Waller 1999, Keeley 2004). In contrast, increased graminoid standing crop associated with native seeded perennials will likely persist into subsequent years, contributing to belowground biomass (Gill et al. 2002). Seeding native perennial species has been promoted as an alternative to annuals to encourage longer-term ecosystem rehabilitation (Richards et al. 1998, Beschta et al. 2003). Since the risk of soil erosion following fire is the greatest in the first few growing seasons a successful treatment would need to produce high amounts of standing crop quickly following treatment application (Wagenbrenner et al. 2006). Neither native nor non-native seeding treatments resulted in higher amounts of total standing crop in the first year.

The minimal effectiveness of seeding to increase vegetative cover or standing crop that we found may be linked to timing and intensity of precipitation (Robichaud et al. 2000). High intensity rains can physically remove and transport surface materials

including seeds (Johansen et al. 2001). High intensity storms have been linked to low success of seeding in previous research (Robichaud et al. 2006, Wabenbrenner et al. 2006). In the mountainous western United States, such high intensity rainstorms commonly coincide or immediately follow mid-summer fires (Swetnam and Betancourt 1990). At all of our study sites, the onset of high intensity summer rainstorms coincided with fire containment, plot establishment, and seeding.

Another factor that may have limited the success of some of our seeded native species is the origin of the seed. Four of the five seeded species were from regionally local sources; however, the squirreltail that comprised almost half of the seed mix was from a non-local source and from unidentified genetic origin. There is evidence, both at small and regional scales, showing many species are genetically adapted to site-specific conditions (Grant 1982, Millar and Libby 1989, Raabova et al. 2007). The risk of seeding with native species that are not locally adapted is twofold. First, the seed may be poorly adapted to site specific conditions, and germination and success of seeding efforts may be reduced (Raabova et al. 2007). Second, gene flow from introduced plants may swamp local populations, effectively removing or altering patterns of local adaptation (Jones 1998, McKay et al. 2005). The degree to which non-local genetics of the native seeded species affected our results is unknown, but could have lasting consequences.

Does seeding reduce non-native species?

We attribute the ineffectiveness of seeding in altering the abundance or richness of non-native species in the first year to two factors. First, non-native abundance and richness were low on all three fires. Second, even on the most successful seeding treatments on the Warm Fire, resulting plant cover was inadequate to exclude non-natives

(Grace 1999, Grime 2001). Species exclusions are generally reported when the seeded species have high cover values or have high dominance in the plant community (Barclay et al. 2004, Keeley 2004). At our two fires where non-natives occurred at higher than trace amounts, seeded wheat established <3% cover and seeded native perennial species provided between 4% and 14% cover. Thus, our plots likely had seeded species abundance values too low to influence non-native abundance. Thompson et al. (2006) experimented with seeding native perennials and found lower non-native annuals on native seeded plots in the third growing season post-burn, indicating the potential longer-term benefits of seeding with native perennial species.

The initial determination of whether to seed or not to seed to control non-native species following a high-severity wildfire is often based on the assumption that non-natives will increase following fire; however, this may not always be the case. Recent studies show two distinctly different patterns regarding non-native species invasions following high-severity fires in ponderosa pine forests. The first pattern indicates colonizing species, including non-natives, increase after high-severity fires and that these increases are correlated with increasing fire severity (Crawford et al. 2001, Griffis et al. 2001, Hunter et al. 2006). These studies indicate that colonizing non-native forb species are particularly abundant in the first few growing seasons following high-severity fires (Crawford et al. 2001, Griffis et al. 2001, Hunter et al. 2006). In contrast, the second pattern indicates non-natives do not increase following fire (Huisinga et al. 2005, Kuenzi et al. 2008). Huisinga et al. (2005) attributed low non-native species abundance in a National Park to low levels of anthropogenic influence, including minimal livestock grazing, logging, and fire suppression. The implication is that risk of introduction has

been low, resulting in low propagule pressure. However, our three study sites and those considered by Kuenzi et al. (2008) have a long history of active management and resource utilization, and both of these studies documented low levels of non-native species following high-severity wildfires.

Some research has indicated the risk of non-native species invasion following high-severity disturbances may increase over time (Keeley 2004). If this is true for semi-arid ponderosa pine forests, seeding with ephemeral non-native species would not be an effective approach for addressing invasions in future growing seasons, as timing of the greatest abundance of seeded species would not coincide with non-native species increases. Some researchers have demonstrated longer-lived perennial species may reduce the abundance of non-native species in the third growing season and beyond (Thompson et al. 2006); implying seeding with native perennials may be a more appropriate method of addressing concerns of non-native species invasions. However, studies that have examined non-native species abundances in the second or third year following high-severity wildfires in ponderosa pine forests do not demonstrate increases in these plants (Huisinga et al. 2005, Kuenzi et al. 2008).

Although the abundance and richness of non-natives is so far low at our study locations, some of the non-native species present are considered “transformer” species and may increase, or have negative ecological effects, over time (Richardson et al. 2000). Of the 11 non-native species we recorded, six are listed as noxious in at least one southwestern state (USDA NRCS 2009, Invaders database 2009). These include cheatgrass, bull thistle, redstem stork’s bill, Dalmation toadflax, little hogweed, and common mullein. Some of these species have been shown to significantly transform

ecosystem structure or function in other forested systems (D'Antonio et al. 1992, Oliff et al. 2001, D'Antonio et al. 2004). Longer term monitoring of these sites is warranted to both assess population trajectories of these non-native species and to better evaluate the relative effectiveness of seeding treatments in restoring native plant communities.

Conclusion

The efficacy of post-wildfire seeding to increase vegetative cover, reduce bare ground, and prevent non-native species invasion following fire is not well supported by previous research (Robichaud et al. 2000). Here we provide evidence for low vegetative cover and standing crop from seeded species in ponderosa pine forests using a replicated experimental design with both native and non-native seeding treatments.

Seeding has been shown to have negative effects on plant communities, which may have long-term negative effects on plant regeneration. Beyers' (2004) review noted that in the cases when seeding provided cover values thought to significantly reduce erosion, there were negative ecological effects as a result of the high abundance of seeded species. Some of these negative effects included reduced abundance of conifer, shrub, and annual colonizing species (Schoennagel and Waller 1999, Keeley 2004). This pattern of dominant species reducing abundance of other species in plant communities is well documented (Stohlgren et al. 2003, Smith et al. 2004, Spehn et al. 2004) and in this context is counter to long-term goals of ecosystem rehabilitation (Keeley 2004). Thus, the paradox of seeding following fires is that seeded species need to achieve high amounts of cover to effectively reduce bare ground and non-native species invasions, but high cover and dominance of seeded species is often associated with negative impacts to plant community regeneration (Beyers 2004, Keeley 2004).

The high financial cost and low potential for effectiveness should call into question the continued practice of seeding areas burned in high-severity wildfires. Robichaud et al. (2000) estimated that post-wildfire rehabilitation over the last decade cost $48 million and Wolfson et al. (2006) reported $6 million was spent in Arizona and New Mexico over a two-year period. Both of these reviews indicate that seeding was the most widely applied post-wildfire treatment. Our study adds to the growing evidence that post-fire seeding is often ineffective in enhancing post-fire plant cover, reducing bare ground, or reducing invasive non-natives.

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Table 1. Seeding treatment species, average pure live seeds (PLS)/kg, target application rates (seeds/m^2), and percentage of species in seed mixes.

#	Treatment type	Species	Target application rate (seeds/m^2)	% of mix	Seed origin
1	Annual ryegrass	*Lolium multiflorum*	403	100%	Field grown, Utah
2	Common wheat	*Triticum aestivum*	403	100%	Field grown, Idaho
3	Native species	Native seed mix	403 Combined		
		Elymus elymoides	173	43%	Field grown, Washington
		Bouteloua gracilis	80	19.8%	Collected, Colorado Plateau
		Poa fendleriana	50	12.4%	Collected, Colorado
		Ipomopsis aggregata	50	12.4%	Collected, Colorado Plateau
		Oxytropis lambertii	50	12.4%	Collected, Colorado Plateau
4	Unseeded	none	0		

Table 2. Average (standard error) percent plant canopy cover of graminoids and forbs, plus cover of seeded species by treatment in the first year post-burn. 't' = <0.5 % cover. A) Warm Fire, B) Potato Fire, C) Birdie Fire.

A) Mean Plant Canopy Cover (%) Seeded Species, Warm Fire

Treatment	Unseeded	Native	Wheat	Ryegrass
Species				
Total Graminoid Cover	**12.79 (1.7)**	**23.97 (2.43)**	**27.11 (3.5)**	**33.29 (3.92)**
Bouteloua gracilis	0.79 (0.39)	0.88 (0.22)	0.65 (0.35)	0.38 (0.22)
Elymus elymoides	3.15 (1.29)	11.64 (2.32)	1.75 (0.58)	0.92 (0.25)
Poa fendleriana	4.73 (0.96)	7.66 (1.78)	3.71 (0.99)	5.18 (1.16)
Lolium multiflorum				29.06 (3.47)
Triticum aestivum			20.71 (3.38)	
Total Forb Cover	**27.07 (3.58)**	**29.51 (3.14)**	**23.09 (2.84)**	**22.49 (2.7)**
Ipomopsis aggregata	t	3.14 (0.13)	t	t
Oxytropis lambertii	t	0.7 (0.05)	t	t

B) Mean Plant Canopy Cover (%) Seeded Species, Potato Fire

Treatment	Unseeded	Native	Wheat
Species			
Total Graminoid Cover	**12.04 (2.53)**	**13.5 (1.51)**	**8.58 (1.43)**
Bouteloua gracilis	1.19 (0.92)	2.84 (0.74)	0.68 (0.30)
Elymus elymoides	0.13 (0.09)	2.89 (2.32)	0.04 (0.25)
Poa fendleriana	0.45 (0.19)	0.15 (0.05)	0.25 (0.18)
Triticum aestivum			1.75 (0.25)
Total Forbs	**27.4 (2.75)**	**31.67 (2.84)**	**29.5 (3.25)**
Ipomopsis aggregata	t	1.15 (0.94)	t
Oxytropis lambertii	t	0.5 (0.05)	t

C) Mean Plant Canopy Cover (%) Seeded Species, Birdie Fire

Treatment	Unseeded	Native	Wheat
Species			
Total Graminoid Cover	**16.88 (3.39)**	**29.86 (4.89)**	**16.19 (3.55)**
Bouteloua gracilis	0.75 (0.46)	9.83 (2.37)	0.15 (0.11)
Elymus elymoides	3.35 (1.71)	5.26 (1.64)	1.29 (0.80)
Poa fendleriana	t	t	t
Triticum aestivum			2.06 (1.19)
Total Forbs	**23.04 (3.8)**	**16.71 (3.36)**	**15.49 (3.55)**
Ipomopsis aggregata	t	t	t
Oxytropis lambertii	t	1.14 (0.34)	t

Table 3.Total plant cover on seeded and unseeded treatments and plant cover of seeded species in the first year following high-severity wildfires from 18 unpublished and published papers, by location and plant community type. The first nine were originally reported in Robichaud et al. (2000) and last nine studies were either not published in the original review or published since the original review. Cover values are treatment averages if given, otherwise ranges (low – high) are shown. Statistical difference between treatments, when reported is signified by (*).

Study Location	Vegetation Type	Treatment Type	Seeded SpeciesCover / Total Cover	Unseeded Control % Cover	Source
Siskiyou Mts. Or.	Douglas-fir	annual ryegrass	49 / 50*	9	Amaranthus 1989
Siskiyou Mts. Or.	Douglas-fir	annual ryegrass	85.2 / 87.1*	23.6	Amaranthus 1989
near Loman, Id.	Douglas-fir	non-native mix[a]	1 / 14	15	Geier-Hayes 1997
near Loman, Id.	ponderosa pine	non-native mix[a]	3 / 10	nd	Geier-Hayes 1997
near Loman, Id.	subalpine fir	non-native mix[a]	nd / 7	nd	Geier-Hayes 1997
near Greenville, Ca.	mixed conifer	non-native mix[b]	nd / 6	7	Roby 1989
Entiat Exp. Forest, Wa.	ponderosa pine - Douglas-fir	non-native mix[c]	3.3 / 10.3	5.6	Tiedemann and Klock 1973
Snow Basin, Or.	pine-mixed fir	non-native mix[d]	12 / 44*	12	Anderson and Brooks 1975
Santa Lucia Mts., Ca.	sugar pine	annual ryegrass	5 to 70 / 10 to 75	5	Griffin 1982
Greenville, Ca.	mixed conifer	non-native mix[a]	10 / 24	27	Ruby 1989
Pocatello, Id.	sagebrush-juniper-grassland	non-native mix[b]	nd / 12.8	18.3	Ratzlaff and Anderson 1995
Eastern Cascades, Wa.	grand fir (Abies grandis)	non-native mix	nd / 41.5	48	Schoennagel and Waller 1999
N.Western Ca. (Megram Fire)	mixed-conifer	barely + mulch	nd / 31.1	24.3	Kruse et al. 2004
Jemez Mtns., NM.	ponderosa pine	annual ryegrass	nd / 16.7	37.5	Barclay et al. 2004
Central Sierras, Ca. (highway fire)	mixed ponderosa pine -oak-	common wheat	67 / 95*	55	Keeley 2004
N.Central Wa. (North 25 Fire)	mixed-conifer	common wheat	4.5 / 18	18	Robichaud et al. 2006
Tintic Valley, Ut.	pinyon juniper	non-native mix[c]	4 / 4.3	7	Thompson et al. 2006
Loveland, Co.	ponderosa pine	non-native mix[e]	nd / 28	43	Wagenbrenner et al. 2006

Seed Mixes: [a]Orchardgrass, tall fescue, timothy, slender wheatgrass, [b]intermediate wheatgrass, Siberian wheatgrass, medacago sativa, [c]crested wheatgrass, Russian wildrye, tall wheatgrass, western wheatgrass, [d]Great Basin wildrye, bluebunch wheatgrass, Indian ricegrass, squirreltail, western wheatgrass, [e]annual cereal grains

Figure 1. Location of study sites. Warm Fire 23,702 ha, Birdie Fire 2,180 ha, and Potato Fire 7,200 ha.

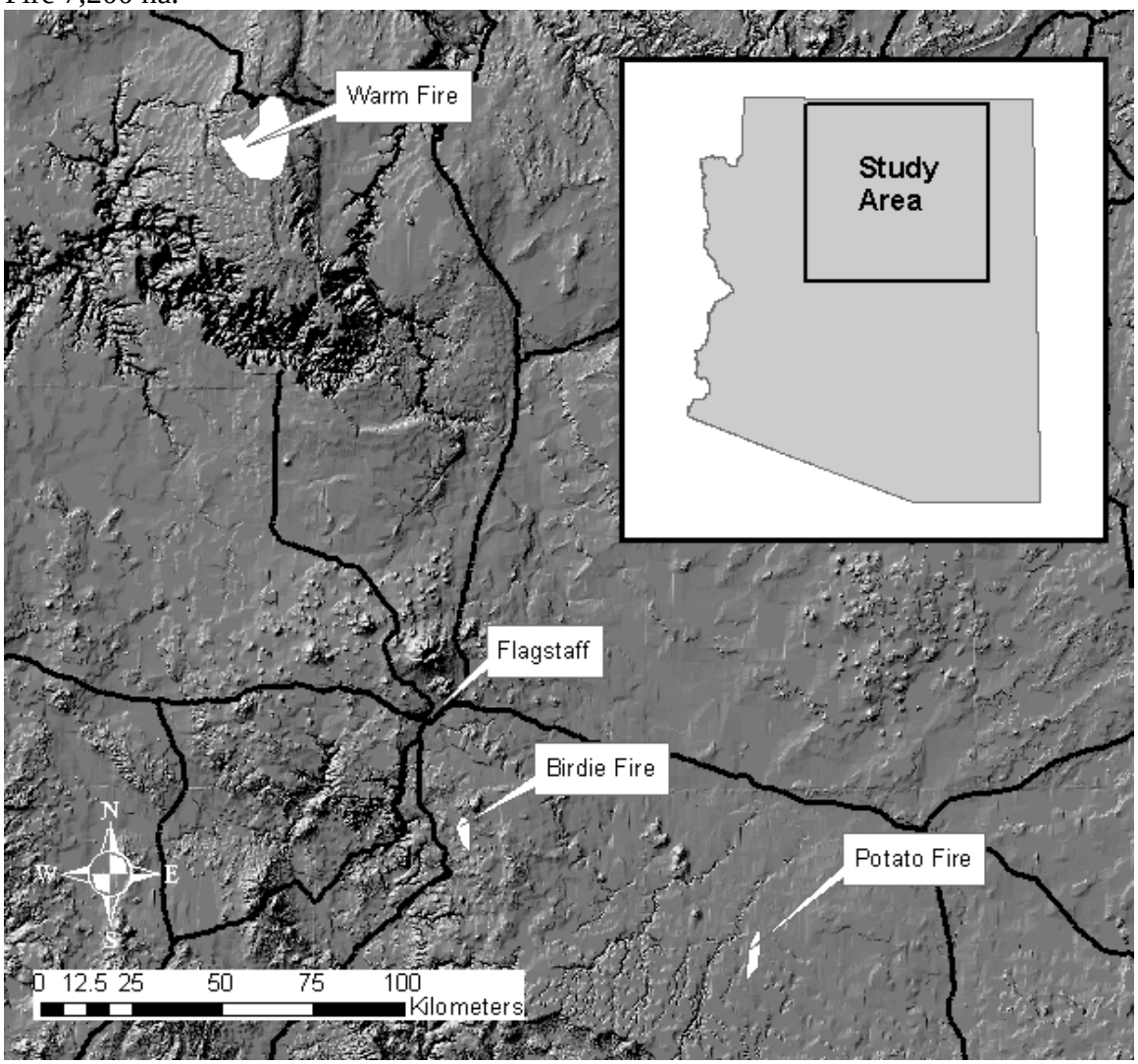


Figure 2. 30-year average precipitation and precipitation for study sites during sampling years. Vertical dashed line is the approximate date of fire containment and date of application of seeding treatments. A) Warm Fire, B) Potato Fire, C) Birdie Fire.

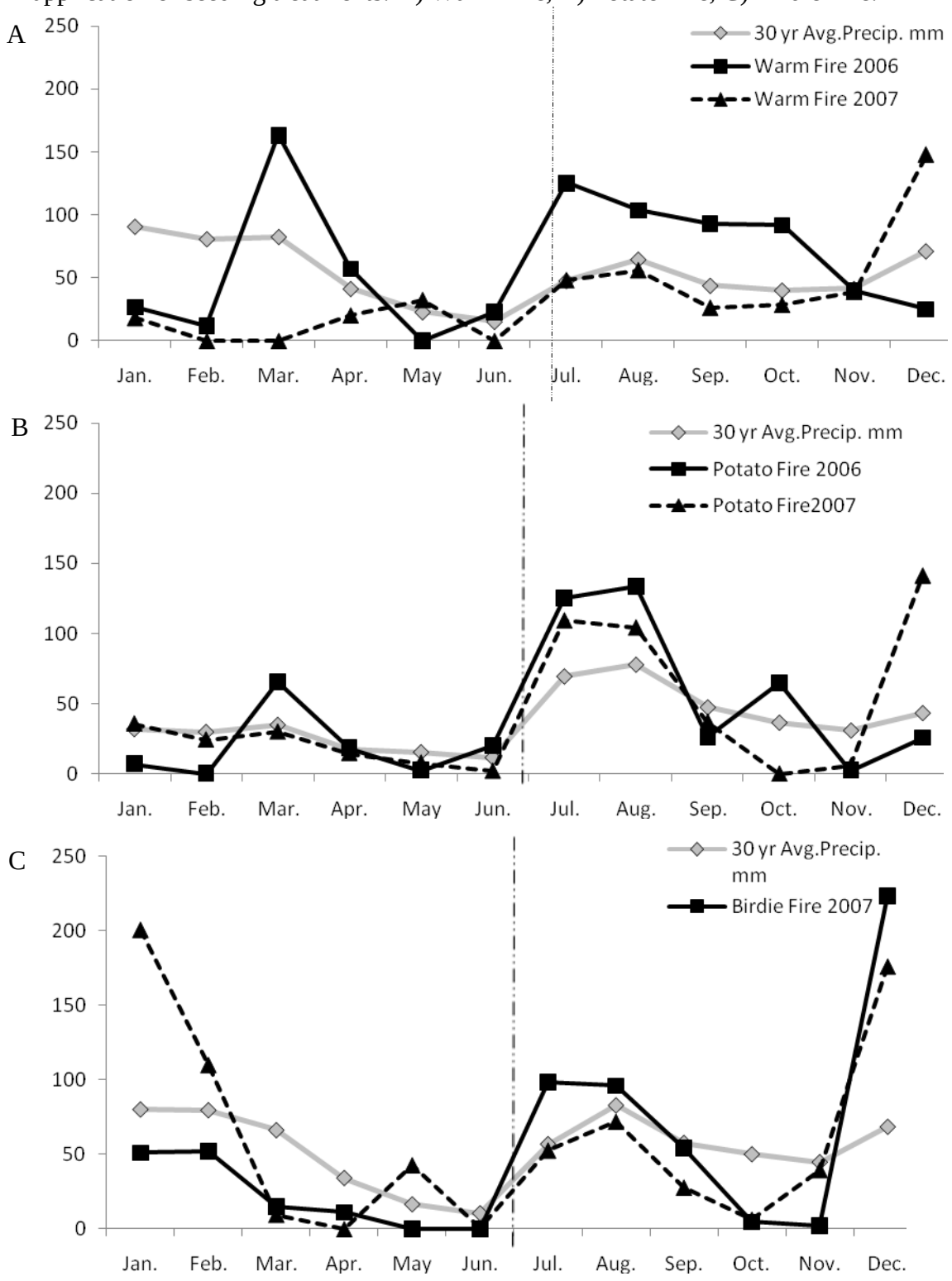


Figure 3. A) Bare ground first year following fire; area between dashed lines is range of bare ground values above which runoff and erosion significantly increase (Johansen et al. 2001). B) Total canopy cover in the first year following fire, dashed line is target cover value associated with lower risk of increased erosion and runoff (Robichaud et al. 2000). Different letters above means indicate significant differences among treatments when they were detected.

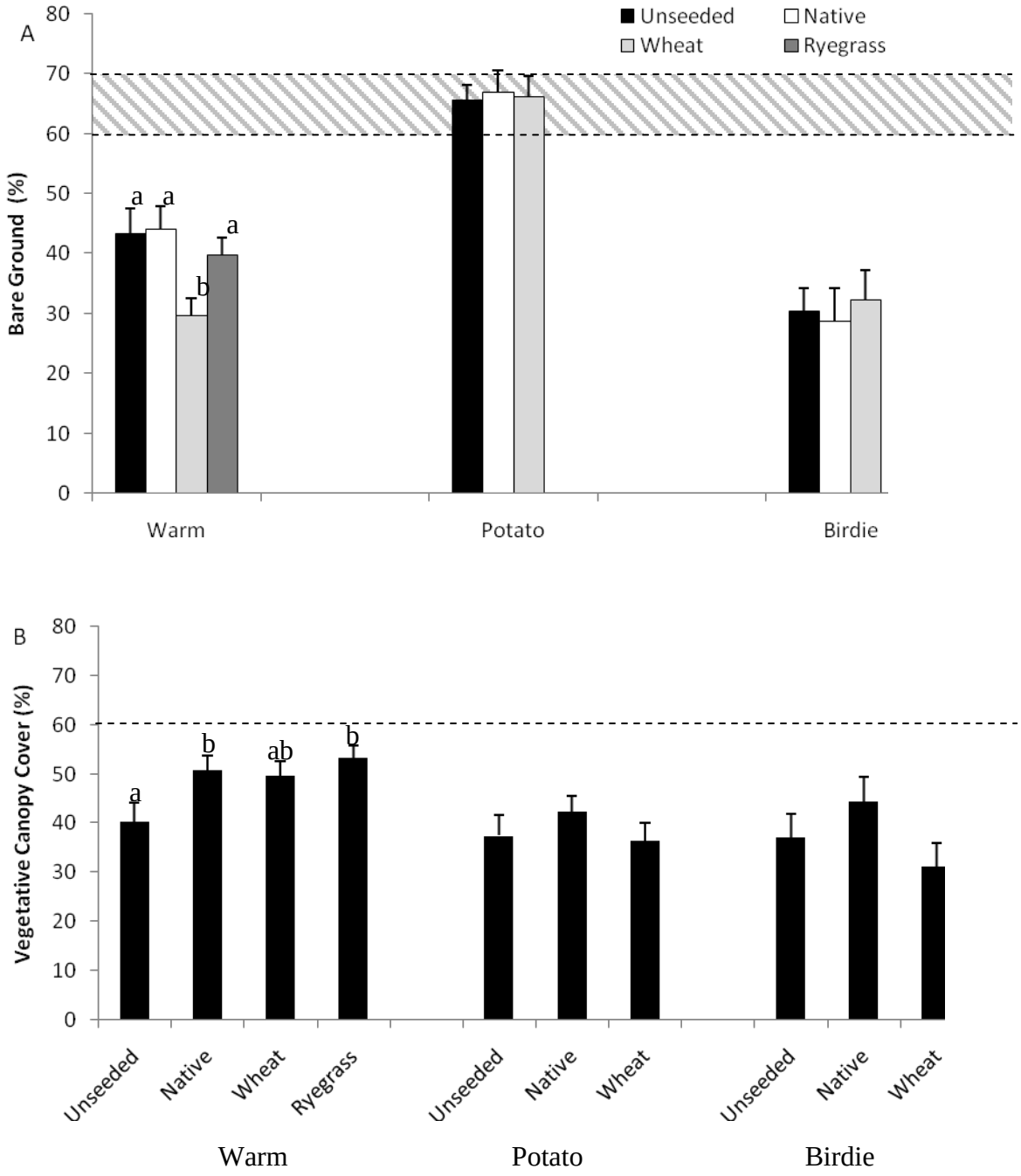
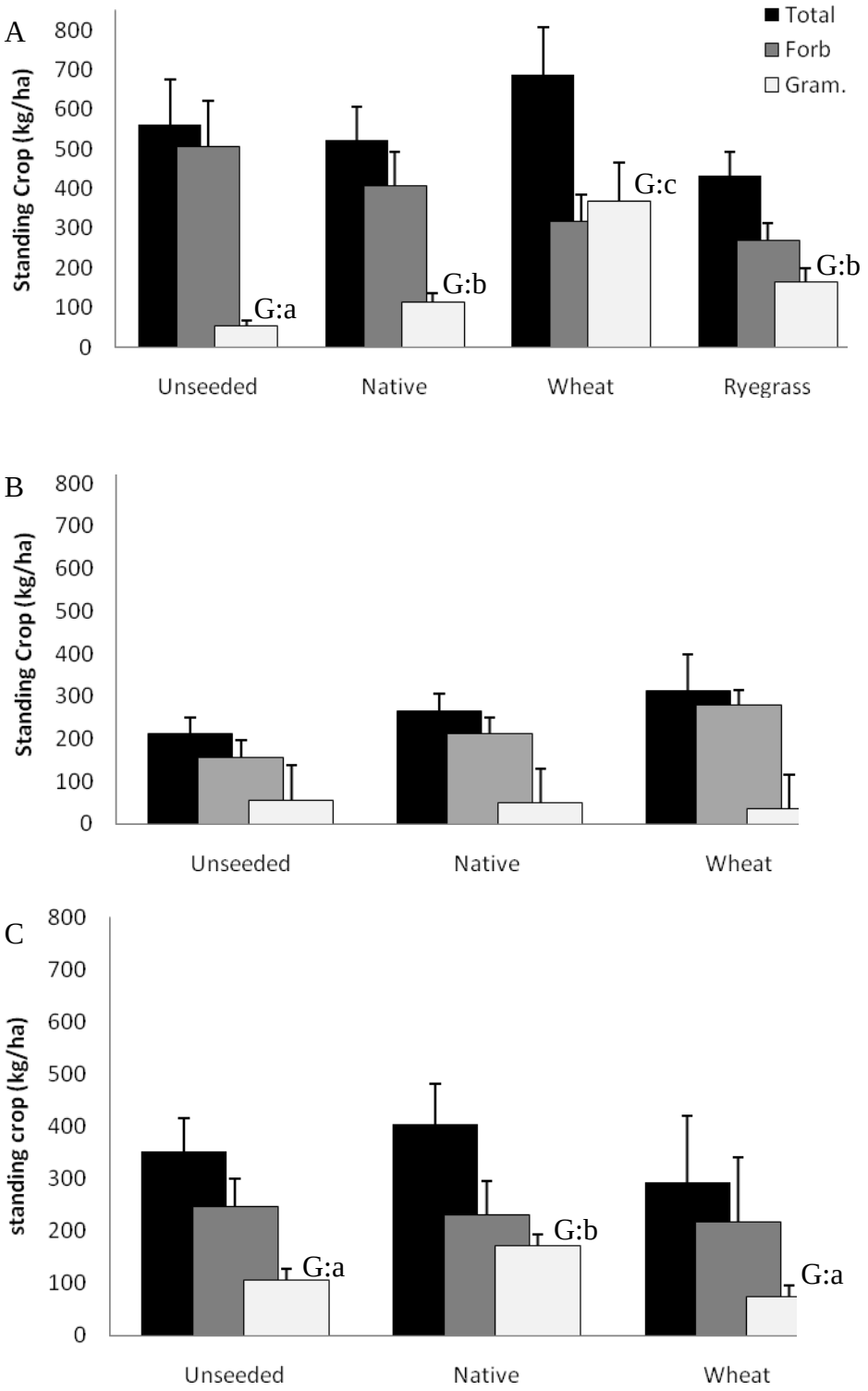


Figure 4. Total standing crop and standing crop by life from group: A) Warm Fire, B) Potato Fire, C) Birdie Fire. Different lower case letters above means indicate significant differences among treatments, upper case letters note life-form groups (G = graminoids).



CHAPTER FOUR

Post-wildfire seeding alters plant community composition

Abstract

Post-wildfire seeding is intended to enhance vegetative cover and thus increase soil water infiltration and reduce erosion. However, successful establishment of seeded species may alter post-fire recovery of native species. We used a replicated and randomized experimental design to determine the effects of seeding on plant communities following two high-severity wildfires in Arizona ponderosa pine (*Pinus ponderosa*) forests. Our treatments included seeding with non-native cereal grains (annual ryegrass, *Lolium multiflorum* (Lam.) and wheat, *Triticum aestivum* L.), seeding with a native seed mix (squirreltail (*Elymus elymoides* (Raf.) Swezey), blue grama, (*Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths), muttongrass (*Poa fendleriana*, (Steud.) Vasey), scarlet gilia (*Ipomopsis aggregata* (Pursh) V.E. Grant), and purple locoweed (*Oxytropis lambertii*, Pursh.)), and an unseeded control. We hypothesized that seeding with native or non-native species would: 1) reduce total species richness and 2) significantly alter plant community composition, and that 3) these changes would persist into the second year following fire occurance. We found that when seeded species accounted for as little as 10% of total plant cover, cover of annual/biennial forbs as a group was significantly reduced and graminoid cover was increased compared to unseeded treatments. Significant reductions in cover of individual forbs, perennials, and a shrub species on seeded treatments compared to unseeded treatments persisted into the second year following fire. The long-term intent of post-fire seeding is to promote ecosystem rehabilitation; thus, alterations to plant community composition may be contrary to these

goals. Further study would be useful in determining the long-term impact of the observations we documented.

Key words: seeding, ponderosa pine, and Burned Area Emergency Rehabilitation

Introduction

In an attempt to ameliorate negative ecological effects of high-severity wildfires, land managers routinely apply post-wildfire treatments, the most common of which is seeding (Robichaud et al. 2000). Post-wildfire seeding is intended to quickly reestablish vegetative cover, reduce bare ground and thereby increase water infiltration into the soil surface and reduce soil erosion (Benavides-Solorio and MacDonald 2001, Johansen et al. 2001). Plant species selected for post-fire seeding treatments are most commonly annual cereal grains or sterile graminoid hybrids selected for quick growth and extensive fibrous roots (Robichaud et al. 2000). Seeding with native species is encouraged but due to low availability of locally collected seed stock and high cost its use is limited (Richards et al. 1998, Beschta et al. 2003).

Post-wildfire seeding has been criticized for the potential of quick-growing seeded species to garner resources and reduce abundance of native plants (Beschta et al. 2003, Keeley 2004). Species with rapid growth, those selected for post-wildfire seeding, have the potential to out-compete and reduce abundance of slower growing species (Corbin and D'Antonio 2004). Non-native cereal grains used in post-fire seeding use nutrient, water, and light resources at higher rates than native species, and may result in lower native species abundance (Keeley 2004, Hunter and Omi 2006). Since post-fire seeding treatments are intended to promote long-term ecosystem rehabilitation, decreasing the abundance of long-lived native perennials and shrubs is of particular concern (Robichaud et al. 2000, Beyers 2004).

Previous research indicates post-wildfire seeding can reduce species richness and abundance of some life-form groups such as annual and biennial forbs, shrubs, and

conifers which may have short and long-term ecological impacts (Schoennagel and Waller 1999, Kruse et al. 2004, Keeley 2004). Annual and biennial forb species are often important contributors to post-fire plant communities and many are uniquely adapted to or dependent on post-fire conditions (Keeley 2004, Laughlin et al. 2004). These pioneer species can provide rapid vegetative cover and thus contribute to reduced erosion and runoff after high-severity fires (Johansen et al. 2001, Laughlin et al. 2004). Shrubs and conifers provide heterogeneity in root depth, soil stabilization, and long-term ecological rehabilitation following high-severity fires in forested ecosystems. Also these longer lived species may serve the role of "foundation species" in many forested systems (Buonopane 2005, Leniere and Houle 2008). Foundation species are those that have significant influence on ecosystem functioning and composition. Significant reductions in these species could have negative long-term effects on ecosystem functioning and diversity (Leniere and Houle 2008).

There is not a clear threshold at which the abundance of seeded species has negative effects on native plant communities. Plant community ecology studies would predict low species richness and recruitment in an ecosystem with high dominance of a single species (Smith et al. 2004). Based on the concept of high dominance, reduced species richness or a general reduction in plant abundance would be expected when seeded species occupied a large portion of the plant community. In Beyers' (2004) review she reported reduced post-fire native plant regeneration is consistently reported when seeded species result in >30% vegetative cover. However, other studies have indicated reduced shrub, annual forb, and biennial forb abundance when seeded species had relatively low cover values but had high relative abundance in the plant community

(Schoennagel and Waller 1999, Barclay et al. 2004, Keeley 2004). Thus, indicating the importance of both total vegetative cover and relative abundance of seeded species in predicting effects of seeding on plant communities.

We seeded native and non-native species following two high-severity wildfires to determine the effects of seeding on plant community composition in the first and second years following fire. We hypothesized that seeding would: 1) reduce total species richness, 2) significantly alter plant community composition, and that 3) these changes would persist into the second year following fire.

Methods

We established experimental plots on two wildfires that occurred in 2006: the Warm Fire in northern Arizona and the Potato Fire in central Arizona. Both wildfires burned in early summer (June-July) in areas dominated by ponderosa pine (*Pinus ponderosa* C. Lawson). Data were collected in the first two years following fire. The Warm Fire receives greater average annual precipitation (65 cm annual average) and is at a higher elevation (2400 m) than the Potato Fire, which receives an average annual precipitation of 52 cm and is at an elevation of 2100 m (Brewer et al. 1991, Laing et al. 1991, WRCC 2009). The Warm Fire is located on the Kaibab Plateau with soils developed from limestone parent material and the Potato Fire is near the southern most edge of the Colorado Plateau with soils developed from basalt parent material.

We established 15 20- x 25-m (500-m^2) plots per treatment at each fire in unseeded areas of high-severity burn, with slopes between 5 to 25%. Treatments randomly assigned to plots included seeding with: 1) non-native common wheat (*Triticum aestivum* L.), 2) non-native annual ryegrass (*Lolium multiflorum* (Lam.) Husnot), 3) a native seed

mix including 43 % squirreltail (*Elymus elymoides* (Raf.) Swezey), 19.8% blue grama (*Bouteloua gracilis* (Willd. ex Kunth) Lag. ex Griffiths), 12.4% muttongrass (*Poa fendleriana*, (Steud.) Vasey), 12.4% scarlet gilia (*Ipomopsis aggregata* (Pursh) V.E. Grant), and 12.4% purple locoweed (*Oxytropis lambertii*, Pursh.), and 4) unseeded control. Species used in seed mixes were selected because of their broad geographic distribution, have shown promising results in previous research, quick growth attributes, and their common use by land managers. (Robichaud et al. 2000, Elseroad et al. 2003, Springer and Laughlin 2004, Kuenzi et al. 2008). Plant nomenclature and nativity are according to the Plants Database (USDA NRCS 2009). Plant reference specimens are stored at the Ecological Restoration Institute at Northern Arizona University, Flagstaff, Arizona. Due to fire size, treatments varied between study sites. At the Warm Fire all four treatments were tested and at the Potato Fire we applied all except the ryegrass treatment. Our target seeding rates were 403 pure live seeds (PLS)/m^2. This seeding rate is similar to those used in other post-fire seeding experiments (Robichaud et al. 2000, Thompson et al. 2006).

Response Variables

We estimated total plant canopy cover, litter, rock, and amount of bare ground, plus cover by life-form (graminoids, shrubs, and forbs) and by species in fifty, 20- X 50-cm (0.1-m^2) quadrats per plot. In each plot, we placed the quadrats at 1-m intervals on the right side of 5 permanent transects. We estimated cover by classes (1 = <1%, 2 = 1-5%, 3 = 6-25%, 4 = 26-50%, 5 = 51-75%, 6 = 76-95%, 7 = 96-100%) modified from Daubenmire (1959). We added the <1% class based on Bailey and Poulton's (1968)

recommendation to better describe vegetation occurring in very small amounts from 0-1 percent. We used mid-points of each cover class to calculate averages.

Analyses

Due to differences in date of burn and variations in treatments applied we analyzed each fire as a separate case study. We used PERMANOVA, a distance-based nonparametric analysis of variance procedure that can be used with univariate or multivariate datasets, in PC-ORD V.5.1; all tests statistics were based on 9999 permutations of the data (McCune and Mefford 1999, Anderson 2001). When significant differences were detected among treatments we used pair-wise *a posteriori* t-test to separate means. For all tests we used $\alpha = 0.05$ to determine significance. To determine if species richness differed among treatments we analyzed the average number of species in each plot in each year and each fire.

To determine if plant community composition varied among treatments in both the first and second years following fire we used non-metric multidimensional scaling (NMDS) (Kruskal 1964) ordination and indicator species analysis (ISA) (Legendre and Legendre 1998). We used NMDS ordinations to show the distribution of mean treatment composition in species space. Ordinations were conducted using PC-ORD software (PC-ORD V. 5.1), the parameters were set at 40 runs with real data, 50 runs with randomizations, a maximum of 400 iterations per run, and an instability criterion of 0.00001 (McCune and Mefford 1999). When a three dimensional solution was recommended we used the two axes that described the greatest amount of variation. Those axis scores were extracted and a mean value and standard errors were calculated per treatment type per year such that each point plotted in the resultant graph is the mean

location of that treatment (n=15) in species space based on ordination scores (Gundale et al. 2006). We used Pearson's correlation between species abundances and axes to determine which species are most strongly correlated with the ordination axis.

We used indicator species analysis (ISA) to identify species most closely associated with each treatment (PC-ORD V.5.1 McCune and Mefford 1999). ISA values were calculated for each species as relative frequency X percent canopy cover. Species with $p < 0.05$ and an indicator value of >25 were identified as indicator species (Dufrene and Legendre 1997).

To determine if compositional differences persisted into the second year, we used univariate analysis of variance, PERMANOVA, to analyze second year data by plant life-form groups including: total forbs, graminoids, and shrubs. We also tested for changes in cover of individual species. We analyzed cover of species that occurred at ~2% cover in at least two treatments plus cover of seeded species. As forbs were the most abundant life-form we separated them into annual/biennial and perennial groups and analyzed these two groups separately using multivariate PERMANOVA. Annual and biennial forbs are often fire followers in ponderosa pine forests and have been shown to be important in providing quick vegetative cover following disturbance, thus documenting the effects of seeding on this group is particularly important (Laughlin et al. 2004, Kuenzi et al. 2008) (Table 1).

We used the Bray-Curtis distance measure for nonparametric analyses of variance and for NMDS procedures (Ludwig and Reynolds 1988, Faith et al. 1987).

RESULTS

Does seeding alter species richness?

We found no support for our hypothesis that seeding altered species richness in the first or second years following fire at either the Warm Fire or the Potato Fire (P = 0.08, 0.19, 0.21, 0.22 respectively) (Table 1). The majority of species recorded at both fires in both years were forbs with annual and biennial species accounting for approximately half of the forb total (Table 2).

Does seeding alter plant community composition?

Warm Fire

Based on ordinations and indicator species analysis, seeding altered plant community composition in both the first and second years following fire (Table 3, Figure 1A and B). Unseeded plots, in both years following fire, were dominated by annual and biennial species. These plots had no significant indicator species in the first year following fire but two annual forbs were indicators in the second. The native seeded plots always had seeded plants as indicators. Abundance of native seeded species on native treatments increased in the second year as did there dissimilarity with unseeded controls, based on ordinations. In the first year following fire both the non-native seeded treatments were separated in ordination space from the unseeded controls and indicator species were the seeded non-natives, this trend continued into the second year at the ryegrass seeded plots but was ephemeral at the wheat seeded plots.

The NMDS ordinations demonstrated spatial separation between plots in both years (Figure 1A and B). This procedure for both the first and second year data set analysis recommended three dimensional solutions; we show the two axes that

represented the greatest variation in the distance matrix. For the first year post-fire data, the three axes combined explained 72% of the variation (R^2 axis 1 = 0.29, axis 2 = 0.25). Final stress was 17.69, final instability was >0.00001, the final solution was based on 208 iterations. Axis one, the X axis on the ordination graph, is most strongly positively correlated with ryegrass and negatively correlated with wheat. Axis two, the Y axis on the ordination graph, is most strongly positively correlated with ryegrass and negatively correlated with two species in the native seed mix: squirreltail and scarlet gilia (Figure 1). The three axes for the second year post-fire ordination combined explained 82% of the variation (R^2 axis 1(X) = 0.31, axis 2(Y) = 0.29). Final stress was 16.34, final instability was >0.00001, the final solution was based on 303 iterations. Axis one, the X axis on the ordination graph, is most strongly positively correlated with blue grama, a seeded native, and negatively correlated with ryegrass and a ruderal forb, spreading fleabane. Axis two, the Y axis on the ordination graph, is most strongly positively correlated with a ruderal forb, running fleabane and negatively correlated with Wheeler's thistle.

Potato Fire

Similar to the Warm Fire composition based on ordinations and ISA was different among treatments in both the first and second years following fire (Table 3, Figure 1C and D). Native treatments always had native seeded species as indicators but had greater separation in ordination space in the second year when seeded species were at higher cover values. Wheat produced very low cover in the first year following fire and nearly disappeared in the second.

In the first year following fire, annual and biennial forbs dominated the plant community and were most closely associated with the ordination axes. In the second

year, seeded natives increased to >10% cover and were correlated with the ordination axes (Figure 1C and D). The NMDS ordination recommended a three dimensional solution for both the first and second year data sets. The axes for the first year ordination represented 81% of the variation (R^2 axis 1 = 0.33, axis 2 = 0.28). Final stress was 14.02, final instability was <0.00001, and the final solution was based on 132 iterations. Axis one was most strongly positively correlated with spreading fleabane, and negatively correlated with common mullein. Axis two is most strongly positively correlated with three nerve goldenrod, *Solidago velutina* DC., and negatively correlated with a native graminoid, *Cyperus fendlerianus* (Boeckeler). The three axes from the second year ordination explained 86% of the total variation (R^2 axis 1 = 0.47, axis 3 = 0.23), final stress was 12.88, based on 114 iterations. Species correlated with axis one were fetid goosefoot (positively) and blue grama (negatively); common mullein (positively) and ragleaf bahia (*Bahia dissecta* (A. Gray) Britton) (negatively) were correlated with axis two.

Indicator species analysis identified seeded species as indicators of seeded treatments in the first and second years following fire (Table 5). Indicator species for the native seeded plots in both years were two seeded grass species, squirreltail and blue grama. These were the two native grasses seeded at the highest rate. Indicator species at the wheat treatment changed from wheat in the first year to a native ruderal forb, fiddleleaf (*Nama dichotomum* (Ruiz and Pav.) Choisy), as wheat completely disappeared in the second year. There were no significant indicators of unseeded plots in the first year following fire, but a ruderal forb, pygmy bluet (*Houstonia wrightii* A. Gray), was an indicator species in the second year.

Do plant community changes due to seeding persist into the second year following fire?

When seeded species accounted for as little as 10% of total cover, the abundance of life-form groups was altered and included a significant increase in graminoids, a general but non-significant decrease in total forb abundance and a significant decrease in annual and biennial forbs compared to unseeded treatments (Table 4 and 5 Figure 2).

Univariate analysis of second year vegetative cover by major life-form groups showed seeding significantly altered graminoid cover at both the Warm and Potato Fire as compared to unseeded controls (P = 0.005 and 0.04). At both fires native seeded plots had higher graminoid cover compared to all other treatments. Total forb cover was lower on seeded plots where seeded species were >3% cover but did not vary significantly among treatments at either the Warm or Potato Fires (P = 0.59 and 0.08 respectively). Total shrub cover was only at trace amounts at the Potato Fire and at the Warm Fire total shrub cover did not vary significantly among treatments (P = 0.07).

Multivariate analysis of the abundance of annual and biennial forbs showed significant differences among treatments at both fires in the second year following fire occurrence (Potato P = 0.03, Warm P = 0.04) (Table 6). At the Warm Fire these forbs were significantly lower on wheat seeded treatments compared to both the native and unseeded treatments (P = 0.01, and 0.02). At the Potato Fire these forbs were significantly lower on native seeded compared to unseeded treatments (P=0.01).

Univariate analysis of abundant species and seeded species at the Warm Fire showed there was significantly lower cover of a perennial bunchgrass, (squirreltail) and a perennial forb (toadflax penstemon, *Penstemon linariodies* A. Gray) in both non-native seeding treatments as compared to unseeded treatments. There was also significantly

greater cover of spreading fleabane (*Erigeron divergens*) in the wheat treatment as compared to unseeded controls. A resprouting shrub (New Mexican locust, *Robinia neomexicana* A. Gray) was significantly lower on ryegrass seeded plots compared to unseeded treatments. Native seeding resulted in significantly higher amounts of squirreltail and muttongrass compared to other treatments. At the Potato Fire native seeding resulted in significantly lower amounts of a ruderal forb, fetid goosefoot (*Chenopodium graveolens* Willd.) and higher amounts of blue grama and bottlebrush squirreltail compared to both wheat and unseeded treatments.

Discussion

Does seeding reduce species richness?

Contrary to our hypothesis, seeding did not reduce total species richness at either fire we studied. A mechanism driving reduced species richness as a result of seeding likely includes high dominance by seeded species (Grime 2001, Smith et al. 2004). Smith et al. (2004) demonstrated lower species richness as a result of high dominance by one or a few species in the plant community. We found relatively low dominance of seeded species; with no reduction in species richness compared to unseeded treatments. Post-fire seeding studies that have reported reduced species richness have also reported high cover values of seeded species, between 40-67%, and relatively high dominance of the seeded species (Conard et al. 1991, Amaranthus et al. 1993, Schoennagel and Waller 1999, Keeley 2004). However, recent reviews indicate it is rare for post-fire seeding treatments to achieve high cover, above 30% vegetative cover (Robichaud et al. 2000, Beyers 2004). Thus, it may be more important to consider the effects of low cover values of seeded species on plant community richness. A few studies have reported reduced

species richness when vegetative cover of seeded species is <30% and these studies generally reported low overall cover such that seeded species are, in relation to all species, relatively high in abundance in the post-fire environment (Ratzlaff and Anderson 1995, Barclay et al. 2004). This provides support for the concept that plant dominance, even at low total vegetative cover values, is important in determining species richness (Smith et al. 2004).

Reducing species richness in post-fire plant communities may be a goal of some seeding treatments, as one of the stated purposes of seeding following fire is to reduce non-native species invasion (Robichaud et al. 2000, Beyers 2004, USDA 2004). In contrast, reducing post-fire plant recruitment and species richness may contradict the goal of promoting ecosystem rehabilitation (Robichaud et al. 2000). In forested ecosystems many perennial species are important for providing long-term soil stability. The high competitive environment resulting from successful seeding treatments has been shown to reduce conifer, shrub, and perennial grass and forb recruitment following fire (Schoennagel and Waller 1999, Keeley 2004). Determining if reduced species richness is a desirable or undesirable outcome of seeding may depend on the abundance, diversity, and propagule pressure of non-natives in the local extant plant community (Sieg et al. 2003).

Does seeding alter plant community composition?

Plant communities were altered as a result of seeding with non-natives, based on our community analyses (NMDS and ISA); differences in composition were ephemeral on wheat seeding treatments but persistent on ryegrass treatments. Altered plant communities as a result of ryegrass seeding have been commonly observed in post-

wildfire seeding studies (Conard et al. 1991, Amaranthus et al. 1993, Barclay et al. 2004). Some researchers have suggested that ryegrass has allelopathic properties that negatively affect plant colonization (Cubbon 1925, Barclay et al. 2004). Other research has suggested that the rapid utilization of nitrogen by ryegrass may reduce abundance of other species (Barclay et al. 2004).

A foundation of the argument in favor of seeding with native species relies on the concept that seeded natives will promote native plants and disrupt native plant regeneration less than seeded non-native cereal grains (Richards et al. 1998, Beschta et al. 2003). We found seeded native species were consistently indicators of native seeded treatments and generally at higher levels in these treatments, although all seeded native species were also present in unseeded treatments. Introducing seed of perennial grasses into the post-fire plant community moved community composition more quickly to a composition that is identified with later seral stages, that is, those with higher abundance of perennial grasses (Moore et al. 2006). The risk of artificially increasing perennial grass species is that colonizing species will be excluded (Anderson and Brooks 1975). Colonizing species are important components of post-fire plant communities in forested ecosystems, in that they provide quick cover following fire and add to biological diversity (Keeley 2004, Laughlin et al. 2004).

Do plant community changes due to seeding persist into the second year following fire?

Our finding that annual and biennial forbs were significantly reduced as a result of seeding and this reduction persisted is important because this group of species can be important in establishing quick vegetative cover following fire (Crawford et al. 2001, Laughlin et al. 2004, Moore et al. 2006, Kuenzi et al. 2008). Not all annual and biennial

forbs found in these plant communities increase following fire but, this grouping of species has been shown to be useful in documenting plant community changes following fire in ponderosa pine forests and is used here for that reason (Laughlin et al. 2004). The reduction in annual and biennial forbs persisted into the second year following fire when the seeded species were as low as <3% cover. This indicates that the result of impacting first year recruitment is longer term reductions of these species. This is of particular concern as many fire-following forbs establish soil seedbanks in the first and second years after disturbance. These seedbanks are important for vegetation recovery following future disturbance events (Wienk et al. 2004, Korb et al. 2005).

Our finding that seeding reduced the abundance of perennials and one shrub species and that these reductions were persistent into the second year following fire indicates seeding may be contradictory to the long-term goals of ecosystem rehabilitation following fire. The long-term intent of post-wildfire seed additions is to promote ecosystem regeneration and rehabilitation (Robichaud et al. 2000, Beyers 2004). In many ecosystems, "foundation species," are those that have a significant influence on ecosystem functioning and often include long lived species such as shrubs and perennial forbs and grasses (Buonopane 2005, Leniere and Houle 2008). Reducing these species can have negative effects on long-term ecosystem development and composition (Elliot and White 1987, Amaranthus et al. 1993). Previous research has indicated seeding with native and non-natives can result in reduced abundance of foundation species such as shrubs, conifers, and perennials (Amaranthus et al. 1993, Beyers 2004, Keeley 2004).

Conclusion

Our study supported the hypothesis that seeding would alter plant community composition and these changes would persist for at least two years following fire. The typical pattern of post-fire plant community development we documented included many species at low abundance values such that the reductions we documented at the species level, even when significant, were relatively small compared to total cover. Reductions in annual and biennial forbs as a group were greater and may have long-term implications on the ability of native plants to establish quick vegetative cover following fire. Longer term studies would help determine the significance of plant community changes resulting from seeding.

The seeded natives produced only about one third of the cover that seeded non-natives did in the first year following fire. This shifted in the second year where non-natives either were not persistent or decreased to approximately 5% cover and seeded natives accounted for approximately 10% cover. This lag time in native seeded species establishment is well documented in other prescribed fire and forest restoration studies (Springer and Laughlin 2004, Daniels et al. 2008). Since the risk of erosion is highest in the first growing season following fire (Robichaud et al. 2006, Wagenbrenner et al. 2006), native perennial grasses tested in this study appear to be poorly suited for post-wildfire seeding treatments.

The intent of post-fire seeding is to promote quick vegetative cover following fire and is predicated on the assumption that natural regeneration will be insufficient to protect soils (Robichaud et al. 2000). However, the difference in total vegetative cover among treatments we documented was small in both years (13% in the first year and 8%

in the second). This study provided two experimental examples that seeding following wildfires does little to increase vegetative cover and alters plant community composition for at least two years following fire. This study adds to the growing evidence that seeding following high-severity wildfires does little to achieve management goals.

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Table 1. Species richness by life-form for the Warm and Potato Fires in the first and second years following fire.

Fire Year	**Forbs, all**	**Annual/Biennial Forbs**	**Graminoid**	**Shrub**	**Tree**	**Total**	**Range per Treatment**
Warm 2007	67	32	15	5	5	92	33 to 39
2008	69	31	17	5	4	95	34 to 42
Potato 2007	58	26	16	4	5	83	43 to 48
2008	75	29	17	5	5	102	44 to 52

Table 2. Annual and biennial forb species (A/B), classified as non-native (NN), occurring in the second year following fire and analyzed as a group with multivariate PERMANOVA analysis.

Genus species	Common name	A/B	NN	Warm/08	Potato/08
Amaranthus sp. L.	pigweed	A		X	X
Androsace septentrionalis L.	pygmyflower rock jasmine	A		X	
Bahia dissecta (A.Gray) Britton	ragleaf bahia	A/B		X	X
Chamaesyce serpylifolia (Pers.) Small	thymeleaf sandmat	A			X
Chenopodium desiccatum A. Nelson	aridland goosefoot	A		X	
Chenopodium graveolens Willd.	fetid goosefoot	A		X	X
Chenopodium leptophyllum (Moq.) Nutt. Ex S. Watson	narrowleaf goosefoot	A		X	
Chenopodium sp.	goosefoot sp.	A		X	X
Cirsium vulgare (Savi) Ten.	bull thistle	B	NN	X	X
Conyza canadensis (L.) Cronquist	Canandian horseweed	A/B		X	X
Descurainia obtusa (Greene) O.E. Schulz	blunt tansymustard	A/B		X	X
Dyssodia papposa (Vent.) Hitchc.	fetid marigold	A			X
Epilobium brachycarpum C. Presl	tall annual willowherb	A		X	
Erigeron canus A. Gray	hoary fleabane	A		X	
Erigeron colomexicanus A. Nelson	running fleabane	B		X	X
Erigeron divergens Torr.&A.Gray	spreading fleabane	B		X	X
Erigeron eatonii A.Gray	Eaton's fleabane	A		X	X
Erodium cicutarium (L.)L'Her.exAiton	redstem stork's bill	A	NN	X	X
Gayophytum ramosissimum Torr. and A. Gray	pinyon groundsmoke	A		X	
Heliomeris multiflora Nutt.	showy goldeneye	B		X	X
Ipomopsis aggregata (Pursh) V.E. Grant	scarlet gilia	B		X	X
Lactuca serriola L.	prickly lettuce	A	NN	X	
Linum australe A. Heller	southern flax	A		X	X
Lupinus kingii S. Watson	King's lupine	A		X	X
Machaeranthera canescens (Pursh) A.Gray	hoary tansyaster	A		X	
Machaeranthera gracilis (Nutt.)Schinners	slender goldenweed	A			X
Nama dichotomum (Ruiz&Pav.)Choisy	wishbone fiddleleaf	A			X
Nicotiana attenuata Torr. Ex S. Watson	coyote tabacco	A		X	X
Orthocarpus purpureoaalbus A. Gray ex S. Watson	purplewhite owl's-clover	A		X	X

Table 2 (Continued). Annual and biennial forb species (A/B), classified as non-native (NN), occurring in the second year following fire and analyzed as a group with multivariate PERMANOVA analysis.

Genus species	Common name	A/B	NN	Warm/08	Potato/08
Packera multilobata (Torr.&GrayexA.Gray)	lobeleaf grounsel	A		X	X
Packera neomexicana (A.Gray)W.A.Weber	New Mexico groundsel	B			X
Phacelia heterophylla Pursh	varileaf phacelia	B		X	X
Plantago argyrea Morris	saltmeado plaintain	A		X	X
Polygonum douglasii Greene	Douglas' knotweed	A		X	
Portulaca oleracea L	little hogweed	A			X
Pseudognaphalium macounii (Greene) Kartesz	Macoun's cudweed	A		X	X
Solanum triflorum Nutt.	cutleaf nightshade	A		X	X
Taraxacum officinale F.H.Wigg.	common dandelion	A		X	X
Tetraneuris acaulis (Pursh)Greene	Stemless four-nerve daisy	B		X	X
Tragopogon dubius Scop.	yellow salsify	A	NN	X	X
Verbascum thapsus L.	common mullein	B	NN	X	X
Verbena bracteata Cav.exLag.&Rodr.	bigbract verbena	A		X	X

Table 3. Indicator species by year and treatment.

Year/Fire	Treatment	Species	ISA Value	p-Value	Category
Warm 2007	Unseeded	----	----	----	----
	Native Seed	**Elymus elymoides*	67	0.0002	P/G
		**Ipomopsis aggregata*	93	0.0002	B/F
		**Oxytropis lambertii*	93	0.0002	P/F
	Wheat	**Triticum aestivum*	89	0.0002	NN/G
	Ryegrass	**Lolium multiflorum*	100	0.0002	NN/G
	Unseeded	*Chenopodium leptophyllum*	46	0.0382	A/F
2008		*Latuca serriola*	29	0.0436	NN/A/F
	Native Seed	* *Elymus elymoides*	53	0.0002	P/G
		* *Ipomopsis aggregata*	97	0.0002	B/F
		* *Oxytropis lambertii*	93	0.0002	P/F
	Wheat	*Erigeron divergens*	38	0.0108	B/F
	Ryegrass	* *Lolium multiflorum*	87	0.0002	NN/G
Potato 2007	Unseeded	----	----	----	
	Native Seed	* *Elymus elymoides*	75	0.0002	P/G
		**Bouteloua gracilis*	55	0.0002	P/G
	Wheat	----	----	----	----
	Unseeded	*Houstonia wrightii*	94	0.0002	A/B/F
2008					
	Native Seed	* *Elymus elymoides*	91	0.0002	P/G
		* *Bouteloua gracilis*	54	0.0002	P/G
	Wheat	*Nama dichotomum*	56	0.0020	A/F

Category: A = annual, B = biennial, P = perennial, G = graminoid, F = Forb, NN = non-native. * species included in seed mix

Table 4. Warm Fire; Average percent (standard error) cover of major species by life-form and treatment in the second (2008) year post-burn. Species included in one of the seeding treatments are indicated with *.

	Treatment			
Life form Species	Unseeded	Native	Ryegrass	Wheat
Graminoids				
**Bouteloua gracilis*	0.60 (0.59)	0.91 (0.41)	0.25 (0.48)	0.45 (0.91)
Carex occidentalis	2.6 (0.59)	1.58 0.41)	2.81 (0.48	3.43 (0.91)
**Elymus elymoides*	3.92 (1.77)	8.68 (1.64)	1.75 0.43)	2.03 (0.65)
**Poa fendleriana*	7.16 (1.01)	11.17 (2.39)	8.62 (1.46)	4.55 (1.54)
**Lolium multiflorum*	— —	— —	5.55 (1.73)	— —
**Triticum aestivum*	— —	— —	— —	2.25 (0.62)
Forbs				
Artemisia ludoviciana	3.25 (1.34)	1.92 (0.68)	1.37 (0.55)	2.03 (0.67)
Chenopodium leptophyllum	1.40 (0.78)	1.36 (0.17)	2.02 (0.08)	1.9 (0.08)
Cirsium wheeleri	2.12 (0.99)	3.90 (1.54)	2.34 (1.60)	2.62 (1.16)
Erigeron divergens	12.95 (3.19)	10.15 (2.56)	14.37 (3.08)	22.81 (2.90)
Eriogonum racemosum	1.94 (0.50)	2.65 (0.55)	1.38 (0.29)	2.48 (0.57)
Lotus wrightii	1.66 (0.85)	2.45 (1.38)	7.94 (3.15)	3.98 (1.18)
Shrubs				
Ericameria nauseosa	4.12 (0.76)	2.36 (0.46)	2.72 (0.75)	4.51 (0.75)
Robinia neomexicana	2.25 (0.750	1.74 (1.20)	0.45 (0.33)	1.16 (0.72)

Table 5. Potato Fire: Average percent (standard error) cover of major species by life-form and treatment in the second (2008) year post-burn. Species included in one of the seeding treatments are indicated with *.

	Treatment		
Life form Species	Unseeded	Native	Wheat
Graminoids			
* *Bouteloua gracilis*	4.21 (1.27)	10.12 (1.41)	4.57 (1.55)
Cyperus fendlerianus	5.96 (1.29)	3.66 (0.54)	3.88 (0.54)
**Elymus elymoides*	0.19 (0.09)	3.14 (0.56)	0.11 (0.08)
* *Poa fendleriana*	1.03 (0.04)	1.37 (0.65)	0.87 (0.33)
* *Triticum aestivum*	-	-	-
Forbs			
Bahia disecta	0.56 (0.27)	1.78 (0.89)	0.56 (0.26)
Chenopodium graveolens	13.25 (2.23)	9.17 (1.77)	13.75 (0.10)
Erigeron colomexicanus	8.03 (2.29)	4.78 (1.03)	4.88 (2.60)
Erigeron divergens	0.14 (0.19)	0.31 (0.19)	0.39 (0.18)
Verbascum thapsus	1.02 (0.50)	1.19 (0.82)	3.22 (1.79)

Table 6. Multivariate PERMANOVA results for annual/biennial forbs. Test Statistic (Test Stat) is psuedo F or T from PERMANOVA analysis.

Fire	Year 2 Test Stat (p)	Pairwise comparisons	Test Stat (p)
Warm	1.92 (0.04)	Wheat vs Native	1.76 (<0.01)
		Wheat vs Unseeded	1.56 (0.02)
Potato	1.847 (0.02)	Native vs Unseeded	1.72 (0.01)

Figure 1. NMDS ordination graph of treatments (n = 15) (A,B) Warm Fire 1st and 2nd year post-burn and (C,D) Potato Fire 1st and 2nd year post-burn

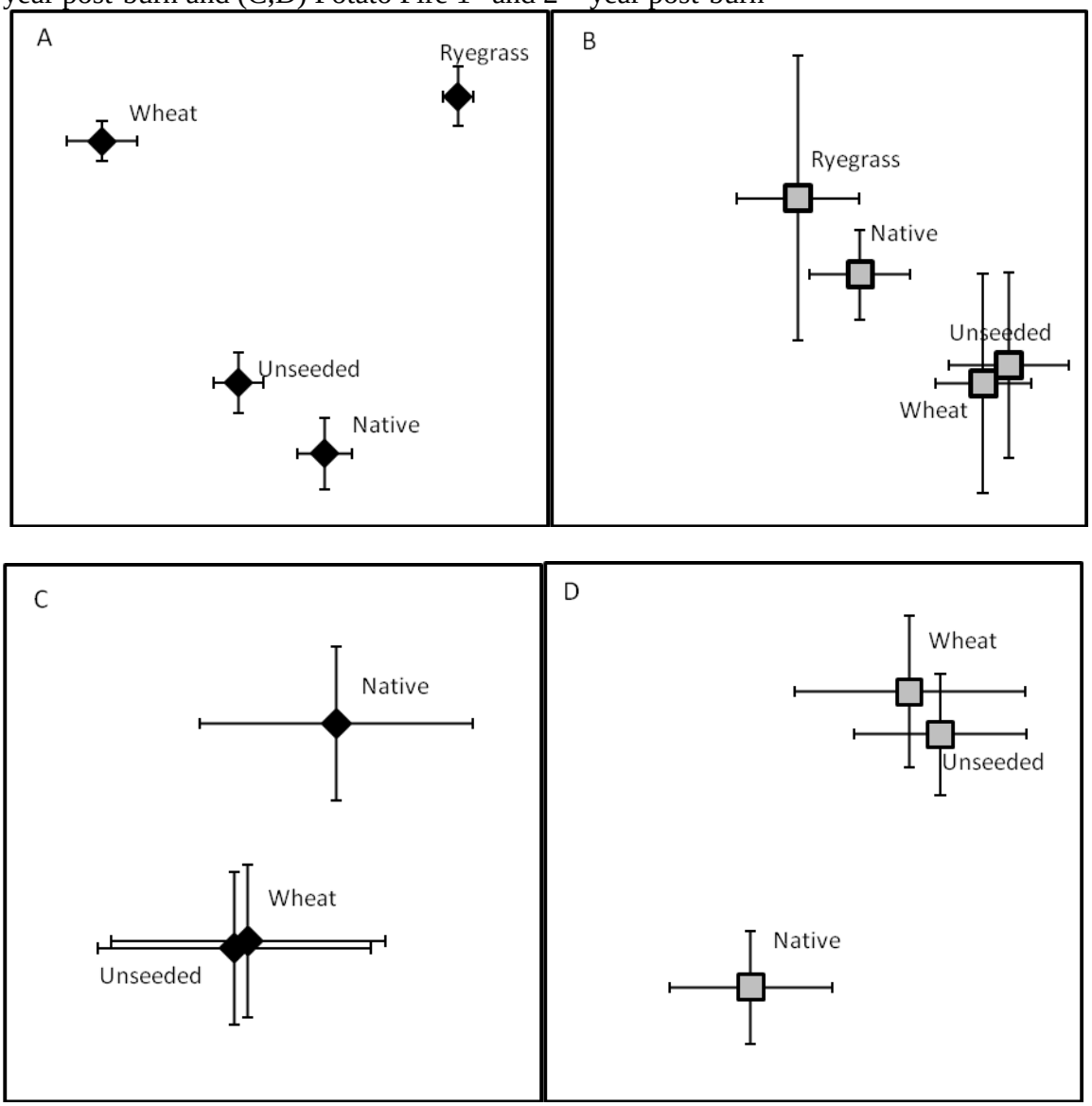
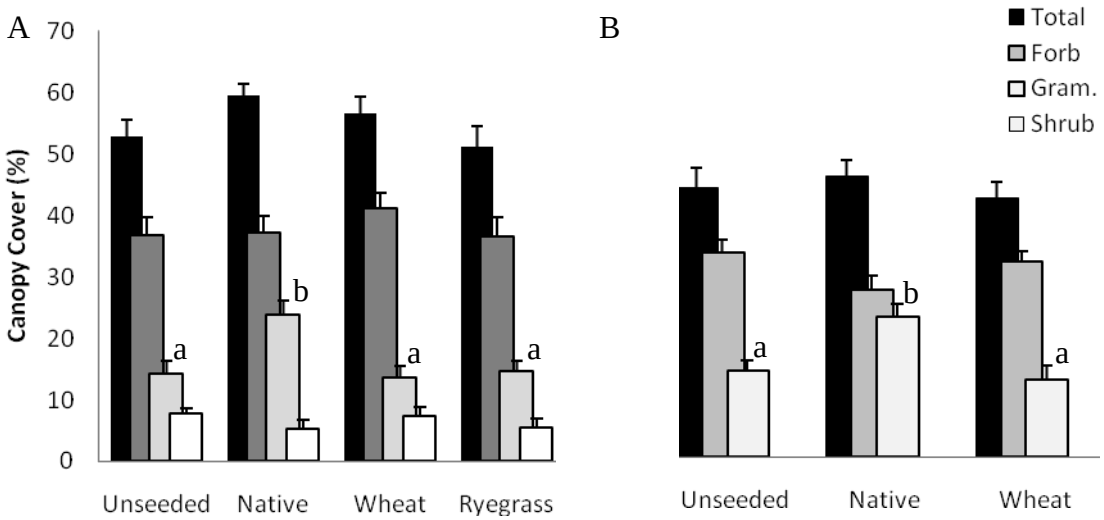


Figure 2. Mean plant canopy cover (%) and standard errors, separated by life forms. Warm Fire (2008) second year post-fire (A), Potato Fire (2008) second year post-fire (B). Significant differences among treatments by life form are indicated with different letters above mean cover.



CHAPTER 5

Conclusion

U.S. federal land managers are directed to evaluate conditions that exist following high-severity wildfires that occur on public land (Robichaud et al. 2000, USDA 2004). These managers evaluate the risk of erosion and non-native species invasion and when it is determined to be great recommend treatments to ameliorate these threats. These actions are presently conducted through the Burned Area Emergency Rehabilitation (BAER) program which in the last two decades spent over 48 million dollars on post-fire rehabilitation projects (Robichaud et al. 2000). Wolfson et al. (2008) estimated the BAER program spent 6 million dollars in Arizona and New Mexico over a recent two year period. BAER teams have a set of actions they commonly recommend, with seeding being the most widely utilized action (Robichaud et al. 2000, USDA 2004). This present study and several preceding it (e.g., Schoennagel and Waller 1999, Kruse et al. 2004) evaluated the effects and effectiveness of post-fire seeding

Using total vegetative cover as a metric of success for seeding treatments, few studies indicate seeding following fires increases area covered by vegetation in the first growing season following fire (Robichaud et al. 2000, Keeley 2004). My study clearly demonstrates the lack of effectiveness of these treatments in increasing vegetative cover following fire. My results support findings of two recent reviews that examined data from published reports and from internal agency monitoring reports that note it is rare for post-fire seeding treatments to result in over 30% vegetative cover in the first growing season following fire (Robichaud et al. 2000, Beyers 2004).

A primary goal of post-fire seeding treatments is to reduce non-native species invasion (Robichaud et al. 2000, USDA 2004). I found no support for the notion that post-fire seeding reduces abundance or richness of non-native species. At two of my study sites I did record relatively high levels of common mullein (*Verbascum thapsus* (L.) and at one site found Kentecky bluegrass (*Poa pratensis* (L.)) and Canadian bluegrass (*Poa compressa* (L.)) to be prevalent. Mullein has a persistent seed bank >200 years and will likely persist at some levels in the plant community of northern Arizona regardless of post wildfire seeding treatments. The non-native grasses I recorded are likely a legacy of past intentional introductions (Fowler et al. 2009). These rhizomatous grasses are also difficult to remove from the plant community and may become a naturalized component of this system.

Seeding treatments, even at low cover values, may have persistent effects on the plant community. It was notable that plant community composition was altered when first year seeded species cover was below 30%. The observation of greater change in ryegrass seeded plots adds to previous research indicating this species may have lasting affects on plant community regeneration (Conard et al. 1991, Barclay et al. 2004). Seeded natives continued to increase in vegetative cover in the second year but never produced high amounts of cover, relative to the total cover. Whether seeding with natives results in communities that are reflective of a later seral stage or represent a significant alteration to plant community composition will require further study. Wheat (*Triticum aestivum* (L.)) produced similar cover values as ryegrass (*Lolium multiflorum* (L.)) but appeared to have less of an impact on the plant community and almost completely disappeared in the second year post-burn. This reduction indicates that wheat

may have the least impact to plant communities, although studies that documented high levels of wheat cover found the persistent thatch layer inhibited plant community regeneration (Keeley 2004).

Although not quantified in my study, it appears that ungulate grazers, such as cattle and elk (*Cervus elaphus*), may be utilizing vegetation in burned areas in the first and second years following fire. At two of my study sites, Potato and Birdie Fires, cattle and elk and their signs were prevalent on all study plots. Plants on study plots had obvious signs of damage from grazing. Other researchers have warned of the negative effects of introducing grazing animals following fire (Beschta et al. 2003). The affect of ungulate grazers on post-fire plant community deserves further research.

Results from this study strongly indicate the ineffectiveness of post-fire seeding to achieve management goals and supports previous research that indicates potential negative impacts to plant communities as a result of post-fire seeding (Schoennagel and Waller 1999, Beyers 2004, Keeley 2004). Seeding following fires has been shown to be an expensive action that yields minimal results (Robichaud et al. 2000). Contour felling of logs following high-severity burns has been shown to have at least limited effectiveness at reducing hill slope erosion and warrants further research (Wagenbrenner et al. 2006). Post-fire mulch additions have also been shown to be effective; however, there is a major concern of non-native species introductions as a result of mulching and further research into this aspect would be useful (Kruse et al. 2004). Based on my results from three well replicated case studies, seeding does not achieve management goals of increasing vegetative cover or reducing non-native species invasions, and contrary to

management goals, negatively impacts native plant regeneration. Managers should consider alternative actions when treating high-severity burn areas.

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APPENDICES

Appendix I. Species list of annual and biennial forbs on the Potato Fire (PTO) in 2007 and 2008 growing seasons; includes all species captured in cover and standing crop sampling. A/B = annual/biennial species, NN = non-native species.

Genus species	Common name	A/B	NN	PTO'07	PTO'08
Amaranthus sp. L.	pigweed	A		X	X
Chamaesyce serpylifolia (Pers.) Small	thymeleaf sandmat	A		X	X
Chenopodium desiccatum A. Nelson	aridland goosefoot	A		X	X
Chenopodium graveolens Willd.	fetid goosefoot	A		X	X
Chenopodium leptophyllum (Moq.) Nutt.	narrowleaf goosefoot	A			X
Chenopodium sp.	goosefoot sp.	A		X	X
Dyssodia papposa (Vent.) Hitchc.	fetid marigold	A			X
Erodium cicutarium (L.)L'Her.exAiton	redstem stork's bill	A	NN	X	X
Lactuca serriola L.	prickly lettuce	A	NN		X
Linum australe A. Heller	southern flax	A		X	X
Lupinus kingii S. Watson	King's lupine	A			X
Machaeranthera gracilis (Nutt.)Schinners	slender goldenweed	A		X	X
Nama dichotomum (Ruiz&Pav.)Choisy	wishbone fiddleleaf	A		X	X
Nicotiana attenuata Torr. Ex S. Watson	coyote tabacco	A			X
Orthocarpus purpureoaalbus A. Gray ex S. Watson	purplewhite owl's-clover	A			X
Packera multilobata (Torr.&GrayexA.Gray)	lobeleaf grounsel	A			X
Plantago argyrea Morris	saltmeado plaintain	A		X	X
Polygonum douglasii Greene	Douglas' knotweed	A		X	X
Portulaca oleracea L.	little hogweed	A		X	x
Pseudognaphalium macounii (Greene) Kartesz	Macoun's cudweed	A		X	X
Solanum triflorum Nutt.	cutleaf nightshade	A		X	X
Taraxacum officinale F.H.Wigg.	common dandelion	A			X
Tragopogon dubius Scop.	yellow salsify	A	NN	X	x
Verbena bracteata Cav.exLag.&Rodr.	bigbract verbena	A			X
Bahia dissecta (A.Gray) Britton	ragleaf bahia	A/B		X	X
Conyza canadensis (L.) Cronquist	Canandian horseweed	A/B		X	
Descurainia obtusa (Greene) O.E. Schulz	blunt tansymustard	A/B		X	X
Hedeoma drummondii Benth.	Drummond's	A/B			X

	fls. pennyroyal				

Appendix I. continued

Genus species	**Common name**	**A/B/P**	**NN**	**PTO'07**	**PTO'08**
Erigeron canus A. Gray	hoary fleabane	A/P		X	
Erigeron eatonii A.Gray	Eaton's fleabane	A/P			X
Cirsium vulgare (Savi) Ten.	bull thistle	B	NN	X	X
Erigeron colomexicanus A. Nelson	running fleabane	B		X	X
Erigeron divergens Torr.&A.Gray	spreading fleabane	B		X	X
Ipomopsis aggregata (Pursh) V.E. Grant	scarlet gilia	B		X	X
Phacelia heterophylla Pursh	varileaf phacelia	B			X
Verbascum thapsus L.	common mullein	B		X	X
Heliomeris multiflora Nutt.	showy goldeneye	B/P		X	X
Packera neomexicana (A.Gray)W.A.Weber	New Mexico groundsel	B/P		X	X
Tetraneuris acaulis (Pursh)Greene	Stemless four-nerve daisy	B/P		X	X

Appendix II. Species list of perennial forbs and succulents on the Potato Fire (PTO) in 2007 and 2008 growing seasons; includes all species captured in cover and standing crop sampling. NN = non-native species.

Genus species	Common name	NN	PTO'07	PTO'08
Antennaria parvifolia Nutt.	small-leaf pussytoes		X	
Arenaria langinosa (Michx.) Rohrb.	spreading sandwort			X
Artemisia carruthii Alph.Wood	Carruth's sagewort		X	X
Artemisia ludoviciana Nutt.	white sagebrush		X	X
Astragalus humistratus A. Gray	groundcover milkvetch			X
Astragalus sp.	milkvetch		X	
Astragalus tephrodes A. Gray	ashen milkvetch		X	X
Calliandra humilis Benth.	dwarf stickpea		X	X
Castilleja linariifolia Benth.	paintbrush		X	X
Chaetopappa ericoides(Torr.) G.L. Nesom	rose heath		X	X
Chamaesyce fendleri (Torr.&A.Gray) Small	Fendler's sandmat		X	X
Cirsium wheeleri (A.Gray) Petr.	Wheeler's thistle			X
Commelina dianthifolia Delile	birdbill dayflower		X	X
Dalea albiflora A. Gray	whiteflower prarie clover		X	X
Dalea purpurea Vent.	purple prairie clover		X	X
Desmanthus cooleyi (Eaton) Trel.	Cooley's bundflower		X	X
Eriogonum elatum Douglas ex Benth.	tall woolly buckwheat		X	X
Eriogonum jamesii Benth.	James' buckwheat			X
Eriogonum racemosum Nutt.	redroot buckwheat		X	X
Gaura cccinea Nutt. Ex Push	scarlet beeblossom		X	
Geranium caespitosum James	geranium			X
Geranium richardsonii Fisch. & Trautv.	Richardson's geranium			X
Helianthella quinquenervis (Hook.) A. Gray	fivenerve helianthella		X	
Houstonia wrightii A. Gray	pygmy bluet		X	X
Hymenopappusfilifolius Hook.	fineleaf hymenopappus		X	X
Ipomoea plummerae A. Gray	morning glory		X	X
Ipomopsis multiflora (Nutt.) V.E. Grant	manyflowered ipomopsis		X	
Lesquerella intermedia (S. Watson)	mid bladderpod		X	X
Lesquerella rectipes Woot.&Standl.	straight bladderpod			
Lesquerella wardii S. Watson	Ward's bladerpod		X	X
Linum lewisii Pursh	Lewis flax		X	
Lithospermum multiflorum Torr. Ex A. Gray	manyflowered stoneseed			X
Lomatium dissectum (Nutt.) Mathias & Constance	fernleaf biscuitroot			X
Lotus wrightii (A. Gray) Greene	Wright's deervetch		X	X
Lupinus argenteus Pursh	silvery lupine			X
Opuntia sp. Mill.	pricklypear		X	

Appendix II.

Genus species	Common name	NN	PTO'07	PTO'08
Oxalis decaphylla Kunth	tenleaf woodsorrel		X	X
Oxytropis lambertii Pursh	purple locoweed		X	X
Penstemon barbatus (Cav.) Roth	beardlip penstemon		X	X
Penstemon linarioides A. Gray	toadflax penstemon		X	X
Penstemon virgatus A. Gray	upright blue beartongue		X	X
Phlox austromontana Coville	mountain phlox		X	X
Pleacanthus spinosus (Nutt.) Rydb.	thorn skeletonweed			X
Potentilla crinita A. Gray	bearded cinquefoil			X
Potentilla hippiana Lehm.	woolly cinquefoil			X
Pseudocymopterus montanus (A.Gray)	alpine false springparsley		X	
Psoralidium tenuiflorum (Pursh) Rydb.	slimflower scurfpea		X	X
Solanum jamesii Torr.	wild potato		X	
Solidago velutina DC.	threenerve goldenrod		X	X
Sphaeralcea parvifolia A. Nelson	smallflower globemallow		X	X
Trifolium pinetorum Greene	woods clover		X	X
Trifolium sp.	clover		X	
Vicia pulchella Kunth	sweetclover vetch		X	

Appendix III. Species list of annual and perennial graminoids, shrubs, and trees, on the Potato Fire (PTO) in 2007 and 2008 growing seasons; includes all species captured in cover and standing crop sampling. A = annual, P = perennial, G = graminoid, S = shrub, T= tree, NN = non-native species.

Genus species	**Common name**	**P**	**G/S/T**	**NN**	**PTO'07**	**PTO'08**
Achnatherum hymenoides Barkworth	ricegrass	P	G		X	X
Aristida arizonica Vasey	Arizona threeawn	P	G			X
Blepharoneuron tricholepis (Torr.) Nas*h*	pine dropseed	P	G			X
Bouteloua curtipendula (Michx.) Torr.	sideoats grama	P	G		X	X
Bouteloua gracilis (Wild.ex Kunth) Lag.	blue grama	P	G		X	X
Carex geophila Mack.	White Mountain sedge	P	G		X	X
Carex occidentalis L.H. Bailey	western sedge	P	G			X
Carex *sp*.	sedge	P	G		X	X
Cyperus fendlerianus Boeckeler	Fendler's flatsedge	P	G		X	X
Elymus elymoides (Raf.) Swezey	squirreltail	P	G		X	X
Koeleria macrantha (Ledeb. Schult.	prairie Junegrass	P	G		X	X
Muhlenbergia minutissima (Steud.) Swallen	annual muhly	A	G			X
Muhlenbergia montana (Nutt.) Hitchc.	mountain muhly	P	G		X	X
Muhlenbertia wrightii Vasey ex J.M. Coult.	spike muhly	P	G		X	X
Poa fendleriana (Steud.) Vassey	muttongrass	P	G		X	X
Schizachyrium scoparium (Michx.) Nash	little bluestem	P	G		X	
Sporobolus cryptandrus (Torr.) A.Gray	sand dropseed	P	G		X	X
Triticum aestivum L.	common wheat	A	G	NN	X	X
Vulpia myuros L.	rat-tail fescue	A	G	NN	X	
Ceanothus fendleri A. Gray	Fendler's ceanothus	P	S		X	X
Gutierrezia sarothrae (Pursh) Britton	broom snakeweed	P	S		X	X
Mahonia repens (Lindl.) G. Don	creeping barberry	P	S		X	X
Juniperus L.	juniper	P	T		X	X

Appendix IV. Species list of annual and biennial forbs on the Warm Fire (WRM) in 2007 and 2008 growing seasons; includes all species captured in cover and standing crop sampling. A = annual , B = biennial, NN = non-native species.

Genus species	Common name	A/B	NN	WRM'07	WRM'08
Amaranthus sp. L.	pigweed	A			X
Androsace septentrionalis L.	pygmyflower rock jasmine	A		X	X
Chamaesyce serpylifolia (Pers.) Small	thymeleaf sandmat	A		X	
Chenopodium desiccatum A. Nelson	aridland goosefoot	A		X	X
Chenopodium graveolens Willd.	fetid goosefoot	A		X	X
Chenopodium leptophyllum (Moq.) Nutt. Ex S. Watson	narrowleaf goosefoot	A		X	X
Chenopodium sp.	goosefoot sp.	A		X	X
Dyssodia papposa (Vent.) Hitchc.	fetid marigold	A			X
Epilobium brachycarpum C. Presl	tall annual willowherb	A			X
Erodium cicutarium (L.)L'Her.exAiton	redstem stork's bill	A	NN		X
Gayophytum ramosissimum Torr. and A. Gray	pinyon groundsmoke	A		X	X
Helianthus annuus L.	common sunflower	A			X
Lactuca serriola L.	prickly lettuce	A	NN	X	X
Lappula occidentalis (S.Watson) Greene	flatspine stickseed	A			X
Linum australe A. Heller	southern flax	A		X	X
Lupinus kingii S. Watson	King's lupine	A		X	X
Machaeranthera canescens (Pursh) A.Gray	hoary tansyaster	A		X	X
Nicotiana attenuata Torr. ex S. Watson	coyote tabacco	A		X	X
Orthocarpus purpureoalbus A. Gray ex S. Watson	purplewhite owl's-clover	A		X	X
Packera multilobata (Torr.& Gray ex A.Gray)	lobeleaf grounsel	A		X	X
Plantago argyrea Morris	saltmeado plaintain	A			X
Polygonum douglasii Greene	Douglas' knotweed	A		X	X
Pseudognaphalium macounii (Greene) Kartesz	Macoun's cudweed	A		X	X
Salsola tragus L.	prickly Russian thistle	A			X
Solanum triflorum Nutt.	cutleaf nightshade	A			X
Taraxacum officinale F.H.Wigg.	common dandelion	A		X	X
Tragopogon dubius Scop.	yellow salsify	A	NN	X	X
Verbena bracteata Cav.ex Lag.& Rodr.	bigbract verbena	A			X
Bahia dissecta (A.Gray) Britton	ragleaf bahia	A/B		X	X
Conyza canadensis (L.) Cronquist	Cnd. horseweed	A/B		X	X

Appendix IV. continued

Genus species	Common name	A/B/P	NN	WRM'07	WRM'08
Descurainia obtusa (Greene) O.E. Schulz	blunt tansymustard	A/B		X	X
Erigeron canus A. Gray	hoary fleabane	A/P			X
Erigeron eatonii A.Gray	Eaton's fleabane	A/P			X
Cirsium vulgare (Savi) Ten.	bull thistle	B	NN	X	X
Erigeron colomexicanus A. Nelson	running fleabane	B		X	X
Erigeron divergens Torr.& A.Gray	spreading fleabane	B		X	X
Erigeron flagellaris A. Gray	trailing fleabane	B			X
Ipomopsis aggregata (Pursh) V.E. Grant	scarlet gilia	B		X	X
Phacelia heterophylla Pursh	varileaf phacelia	B			X
Verbascum thapsus L.	common mullein	B			x
Heliomeris multiflora Nutt.	showy goldeneye	B/P		X	X
Packera neomexicana (A.Gray) W.A.Weber	New Mexico groundsel	B/P			X
Tetraneuris acaulis (Pursh) Greene	Stemless four-nerve daisy	B/P		X	X

Appendix V. Species list of perennial forbs on the Warm Fire (WRM) in 2007 and 2008 growing seasons; all species captured in cover and standing crop sampling. NN = non-native species.

Genus species	Common name	NN	WRM'07	WRM'08
Agoseris sp. Raf.	agoseris		X	X
Anaphalis margaritacea (L.) Benth.	pearly everlasting		X	
Antennaria parvifolia Nutt.	small-leaf pussytoes			X
Antennaria rosea Greene	rosy pussytoes		X	X
Arabis fendleri (S.Watson) Greene	Fendler's rockcress		X	X
Arenaria fendleri A. Gray	Fendler's sandwort		X	X
Artemisia carruthii Alph.Wood ex Carruth	Carruth's sagewort		X	X
Artemisia ludoviciana Nutt.	white sagebrush		X	X
Asclepias asperula (Decne.) Woodson	spider milkweed		X	X
Astragalus argophyllus Nutt.	silverleaf milkvetch		X	X
Astragalus kentrophyta A. Gray	spiny milkvetch			X
Astragalus sp.	milkvetch		X	X
Castilleja linariifolia Benth.	paintbrush		X	X
Cirsium wheeleri (A.Gray) Petr.	Wheeler's thistle		X	X
Dalea albiflora A. Gray	whiteflower prarie clover		X	X
Dalea purpurea Vent.	purple prairie clover			X
Eriogonum elatum Douglas ex Benth.	tall woolly buckwheat			X
Eriogonum racemosum Nutt.	redroot buckwheat		X	X
Euphorbia brachycera Engelm.	horned spurge		X	X
Geranium caespitosum James	pineywoods geranium		X	X
Hymenopappus filifolius Hook.	fineleaf hymenopappus		X	X
Lesquerella intermedia (S. Watson)	mid bladderpod			X
Lesquerella rectipes Woot.& Standl.	straight bladderpod			X
Lesquerella wardii S. Watson	Ward's bladerpod		X	X
Linum lewisii Pursh	Lewis flax		X	X
Lithospermum incisum Lehm.	narrowleaf stoneseed			X
Lithospermum multiflorum Torr. Ex A. Gray	manyflowered stoneseed		X	X
Lomatium foeniculaceum A. Gray	MacDougal's biscuitroot		X	
Lotus wrightii (A. Gray) Greene	Wright's deervetch		X	X
Lupinus argenteus Pursh	silvery lupine		X	X
Eriogonum elatum Douglas	tall woolly			X

ex Benth.	buckwheat

Appendix V. continued

Genus species	Common name	NN	WRM'07	WRM'08
Oxytropis lambertii Pursh	purple locoweed		X	X
Penstemon barbatus (Cav.) Roth	beardlip penstemon		X	X
Penstemon linarioides A. Gray	toadflax penstemon		X	X
Penstemon virgatus A. Gray	upright blue beartongue			X
Phlox austromontana Coville	mountain phlox		X	X
Pleacanthus spinosus (Nutt.) Rydb.	thorn skeletonweed		X	X
Potentilla crinita A. Gray	bearded cinquefoil		X	X
Potentilla hippiana Lehm.	woolly cinquefoil		X	X
Pseudocymopterus montanus (A.Gray)	alpine false springparsley		X	X
Psoralidium tenuiflorum (Pursh) Rydb.	slimflower scurfpea			X
Solidago velutina DC.	threenerve goldenrod		X	X
Thalictrum fendleri Engelm.ex A.Gray	Fendler's meadow-rue		X	X
Trifolium pinetorum Greene	woods clover		X	X

Appendix VI. Species list of annual and perennial graminoids, shrubs, and trees, on the Warm Fire (WRM) in 2007 and 2008 growing seasons; all species captured in cover and standing crop sampling. A = annual P = biennial species, G = graminoid, S = shrub, T = tree, NN = non-native species.

Genus species	Common name	A/P	G/S/T	NN	WRM'07	WRM'08
Achnatherum hymenoides Barkworth	ricegrass	P	G			X
Aristida arizonica Vasey	Arizona threeawn	P	G			X
Blepharoneuron tricholepis (Torr.) Nash	pine dropseed	P	G			X
Bouteloua gracilis (Wild.ex Kunth) Lag.	blue grama	P	G		X	X
Bromus tectorum L.	cheatgrass	A	G	NN	X	X
Carex geophila Mack.	White Mountain sedge	P	G		X	
Carex occidentalis L.H. Bailey	western sedge	P	G		X	X
Carex *sp.*	sedge	P	G			X
Cyperus fendlerianus Boeckeler	Fendler's flatsedge	P	G			X
Elymus elymoides (Raf.) Swezey	squirreltail	P	G		X	X
Hesperostipa comata (Trin. & Rupr.) Barkworth	needle and thread	P	G		X	X
Koeleria macrantha (Ledeb. Schult.	prairie Junegrass	P	G		X	X
Lolium multflorum Lam.	annual ryegrass	A	G	NN	X	X
Muhlenbergia montana (Nutt.) Hitchc.	mountain muhly	P	G		X	X
Poa fendleriana (Steud.) Vassey	muttongrass	P	G		X	X
Sporobolus cryptandrus (Torr.) A.Gray	sand dropseed	P	G		X	X
Triticum aestivum L.	common wheat	A	G	NN	X	X
Vulpia octoflora (Walter) Rydb.	sixweeks fescue	A	G			X
Ceanothus fendleri A. Gray	Fendler's ceanothus	P	S		X	X
Ericameria nauseosa (Pall. Ex Pursh) G.L.	rubber rabbitbrush	P	S		X	X
Gutierrezia sarothrae (Pursh) Britton	broom snakeweed	P	S		X	X
Mahonia repens (Lindl.) G. Don	creeping barberry	P	S		X	X
Ribes L.	currant	P	S		X	X
Robinia neomexicana A. Gray	New Mexico locust	P	S		X	X
Symphoricarpos albus (L.) S.F. Blake	common snowberry	P	S			X
Pinus ponderosa C. Lawson	ponderosa pine	P	T		X	X
Populus tremuloides Michx.	quaking aspen	P	T			X

Appendix VII. Species list of annual and biennial forbs on the Birdie Fire (BRD) in 2008 growing season; all species captured in cover and standing crop sampling. A/B = annual/ biennial species, NN = non-native species.

Genus species	Common name	A/B	NN
Amaranthus sp. L.	pigweed	A	
Chamaesyce serpylifolia (Pers.) Small	thymeleaf sandmat	A	
Chenopodium graveolens Willd.	fetid goosefoot	A	
Chenopodium sp.	goosefoot sp.	A	
Drymaria leptophylla (Cham.& Schltdl.) Fenzl	canyon drmary	A	
Drymaria molluginea Lag.	slimeleag drymary	A	
Dyssodia papposa (Vent.) Hitchc.	fetid marigold	A	
Epilobium brachycarpum C. Presl	tall annual willowherb	A	
Gayophytum ramosissimum Torr. And A. Gray	pinyon groundsmoke	A	
Lactuca serriola L.	prickly lettuce	A	NN
Nama dichotomum (Ruiz&Pav.)Choisy	wishbone fiddleleaf	A	
Packera multilobata (Torr.&GrayexA.Gray)	lobeleaf grounsel	A	
Polygonum aviculare L.	prostrate knotweed	A	NN
Polygonum douglasii Greene	Douglas' knotweed	A	
Portulaca oleracea L.	little hogweed	A	
Pseudognaphalium macounii (Greene Kartesz)	Macoun's cudweed	A	
Silene antirrhina Lehm.	sleepy silene	A	
Solanum triflorum Nutt.	cutleaf nightshade	A	
Taraxacum officinale F.H.Wigg.	common dandelion	A	
Tragopogon dubius Scop.	yellow salsify	A	NN
Bahia dissecta (A.Gray) Britton	ragleaf bahia	A/B	
Coreopsis tinctoria Nutt.	golden tickseed	A/B	
Malva neglecta Wallr.	common mallow	A/B	NN
Cirsium vulgare (Savi) Ten.	bull thistle	B	NN
Erigeron divergens Torr.&A.Gray	spreading fleabane	B	
Ipomopsis aggregata (Pursh) V.E. Grant	scarlet gilia	B	
Pennellia longifolia (Benth.) Rollins	longleaf mock thelypody	B	
Phacelia heterophylla Pursh	varileaf phacelia	B	
Verbascum thapsus L.	common mullein	B	
Heliomeris multiflora Nutt.	showy goldeneye	B/P	
Packera neomexicana (A.Gray)W.A.Weber	New Mexico groundsel	B/P	

Appendix VIII. Species list of perennial forbs on the Birdie Fire (BRD) in 2008 growing season; all species captured in cover and standing crop sampling. NN = non-native species.

Genus species	**Common name**	**NN**
Achillea millefolia L.	common yarrow	
Antennaria parvifolia Nutt.	small-leaf pussytoes	
Antennaria rosea Greene	rosy pussytoes	
Arabis fendleri (S.Watson) Greene	Fendler's rockcress	
Arenaria langinosa (Michx.) Rohrb.	spreading sandwort	
Artemisia carruthii Alph.Wood ex Carruth	Carruth's sagewort	
Artemisia ludoviciana Nutt.	white sagebrush	
Astragalus calycosus Torr.exS. Watson	Torrey's milkvetch	
Astragalus sp.	milkvetch	
Calliandra humilis Benth.	dwarf stickpea	
Cirsium wheeleri (A.Gray) Petr.	Wheeler's thistle	
Cologania angustifolia Kunth	longleaf cologania	
Dalea albiflora A. Gray	whiteflower prarie clover	
Erigeron fomosissimus Greene	beautiful fleabane	
Eriogonum racemosum Nutt.	redroot buckwheat	
Helianthella quinquenervis (Hook.) A. Gray	fivenerve helianthella	
Houstonia wrightii A. Gray	pygmy bluet	
Hymenopappus filifolius Hook.	fineleaf hymenopappus	
Ipomoea plummerae A. Gray	morning glory	
Iris missouriensis Nutt.	Rocky Mountain iris	
Linaria dalmatica (L.) Mill.	Dalmatian toadflax	NN
Lotus wrightii (A. Gray) Greene	Wright's deervetch	
Lupinus argenteus Pursh	silvery lupine	
Oxalis decaphylla Kunth	tenleaf woodsorrel	
Oxytropis lambertii Pursh	purple locoweed	
Penstemon barbatus (Cav.) Roth	beardlip penstemon	
Potentilla crinita A. Gray	bearded cinquefoil	
Pseudocymopterus montanus (A.Gray)	alpine false springparsley	
Solidago velutina DC.	threenerve goldenrod	
Thalictrum fendleri Engelm.exA.Gray	Fendler's meadow-rue	
Trifolium sp.	clover	
Vicia americana Muhl. Ex Wild.	American vetch	
Vicia pulchella Kunth	sweetclover vetch	

Appendix IX. Species list annual and perennial grasses, shrubs, and trees on the Birdie Fire in 2008 growing season; all species captured in cover and standing crop sampling. A/P = annual/perennial species, G = graminoid, S= shrub, T= tree, NN = non-native species.

Genus species	**Common name**	**A/P**	**G/S/T**	**NN**
Blepharoneuron tricholepis (Torr.) Nas*h*	pine dropseed	P	G	
Bouteloua curtipendula (Michx.) Torr.	sideoats grama	P	G	
Bouteloua gracilis (Wild.ex Kunth) Lag.	blue grama	P	G	
Bromus inermis Leyss.	smooth brome	P	G	NN
Bromus tectorum L.	cheatgrass	A	G	NN
Carex geophila Mack.	White Mountain sedge	P	G	
Cyperus fendlerianus Boeckeler	Fendler's flatsedge	P	G	
Elymus elymoides (Raf.) Swezey	squirreltail	P	G	
Festuca arizonica Vasey	Arizona fescue	P	G	
Hordeum jubatum L.	foxtail barley	P	G	
Koeleria macrantha (Ledeb. Schult.	prairie Junegrass	P	G	
Muhlenbergia minutissima (Steud.) Swallen	annual muhly	A	G	
Muhlenbergia montana (Nutt.) Hitchc.	mountain muhly	P	G	
Muhlenbergia racemosa (Michx.) Briton	marsh muhly	P	G	
Muhlenbergia wrightii Vasey ex J.M. Coult.	spike muhly	P	G	
Poa compressa L.	Canada bluegrass	P	G	NN
Poa fendleriana (Steud.) Vassey	muttongrass	P	G	
Poa pratensis L.	Kentucky bluegrass	P	G	
Triticum aestivum L.	common wheat	A	G	NN
Ceanothus fendleri A. Gray	Fendler's ceanothus	P	S	
Mahonia repens (Lindl.)	creeping barberry	P	S	
Robinia neomexicana A. Gray	New Mexico locust	P	S	
Pinus ponderosa C. Lawson	ponderosa pine	P	T	
Quercus gambelii Nutt.	Gambel oak	P	T	