Changes in vegetation and fuels due to the Warm Fire on the Kaibab National Forest
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Study summary

Fire is a significant and essential disturbance in ponderosa pine ecosystems but the management and the re-introduction of fire across the landscape is a difficult task for land managers. In this study we worked with land managers, stakeholders and researchers to examine the effects of a large wildfire on the Kaibab Plateau in northern Arizona. We analyzed litter and duff depth, downed woody debris and understory vegetation responses to low and high burn severity and assessed the response of the understory vegetation to seeding with *Lolium perenne* ssp. *multiflorum* in high severity burn areas. To assist land managers in future decision making we collected and analyzed data on ponderosa pine mortality, and overstory characteristics by fire severity. In addition, we looked for correlations between pre-fire predictions for fire behavior and fire hazard compared to fire severity.

Abstract

Changing climatic conditions coupled with altered forest conditions and fire regimes have resulted in fires outside the historical range of variability. While fire is an essential disturbance in ponderosa pine (*Pinus ponderosa var. scopulorum* Lawson) ecosystems, high severity crown fires as opposed to the historical low severity surface fires, create a vulnerable landscape. The immediate loss of vegetative cover after a fire leads to concerns of soil erosion and invasion by exotic plants and often drives post-fire rehabilitation. For this study we examined three specific topics related to the Warm Fire on the North Kaibab Plateau in Arizona: (A) How fire severity effects the overstory structure, understory vegetation and fuels, (B) Post-fire rehabilitation effects on understory vegetation, and (C) Landscape level fire effects and their relation to pre-fire characteristics.

Overstory structure of ponderosa pine forests has a significant influence on the fire severity as well as being ultimately determined by resulting severity. We observed a strong positive relationship between tree mortality and post-fire burn severity measures. The relationships between post-fire burn severity, tree density and basal area was negative, thus supporting the expectation that stands with larger but fewer trees would have a lower fire severity rating.

We tested the overall response of vegetation to varying fire severities by comparing the effects of both low and high severity fire compared to unburned controls. Native vegetation in the ponderosa pine ecosystems is adapted to the historic fire regime of frequent low severity surface fires. We found areas that had burned in low severity fire responded with increased species richness but no increase in total vegetative cover and were compositionally more similar to unburned controls. High severity fire was strongly correlated with increased species richness, increased annual and biennial forbs, and increased vegetative cover in the understory. This study supports the continuation of reintroducing fire to ponderosa pine forests in northern Arizona as a means to improve forest health and sustainability by altering understory plant communities.

In terms of fuel accumulation, we observed significantly less total debris in both high and low severity when compared to unburned controls. Litter and duff accumulation was notably less in burned areas and surprisingly decreased from 2008 to 2009. Solid 1000 hour fuels

significantly increased in high severity areas and surpassed controls in 2009 which can be attributed to increased downfall.

In addition to the effects of fire severity, we tested the effects and effectiveness of seeding of on understory vegetation. In our study, seeding with *Lolium perenne ssp. multiflorum* (L.) did not provide adequate vegetative cover to decrease soil erosion and was no more effective at preventing exotic invasions than natural recovery. Our results suggest that presence of the seeded species may have led to a decrease of annual and biennial forbs and three native bunchgrasses. The community composition was significantly different between seeded and non-seeded plots in all three years and is changing at a similar rate but possibly in different trajectories. This study adds to the growing body of evidence that post-fire seeding falls short of management goals and may have unintended consequences on the native plant community.

At the landscape level, burn severity maps for the Warm fire were created with both the differenced Normalized Burn Ratio (dNBR) and the relativized differenced Normalized Burn Ratio (RdNBR). Assessments of these burn severity maps were done using the continuous data through correlations to field data and through classified data using accuracy assessments. Overall the dNBR and RdNBR performed well with correlations coefficients of 0.80 and 0.84 respectively. Accuracy assessments showed moderate agreement with the field data and in particular, the RdNBR classified more of the landscape at high severity giving higher users accuracies for that category. The RdNBR burn severity map was then compared to modeled fire behavior and fire hazard values from FlamMap. There were some interesting similarities between the two despite the FlamMap modeling using only 97% weather and consistent winds from the Southwest. This work points to possible relationships between modeled fire hazard values and resultant fire severity values. However, as expected, the issues of wind direction and weather confound the results.

Background

Wildfires are a natural and significant disturbance in ecosystems across the U.S. but alterations to the landscape from humans and changing climatic conditions have reduced the role of fire in shaping these systems (Agee 1998; Westerling et al. 2006). Fire is integral to maintaining the stability and sustainability of forest health and in facilitating the architecture of overstory trees, composition of vegetation communities and nutrient cycling (Cooper 1960; Fulé et al. 1997). Ponderosa pine (*Pinus ponderosa var. scopulorum* C. Laws.) ecosystems in the Southwest are strongly influenced by fire and have been particularly impacted by over a century of fire suppression, grazing and timber extraction (Cooper 1960; Agee 1998). Early descriptions of ponderosa pine forests depict open parks with clusters of large diameter old growth pine trees interspersed with various mixed aged pine trees (Lang and Stewart 1910; Cooper 1960); this is quite contrary to the forests of the American Southwest today that have little understory vegetation and numerous small diameter trees and in general, current forest conditions are vastly different in structure, function and ecological processes (Covington and Moore 1994; Allen et al. 2002). The interactions of historic land use and climate change have resulted in high levels of fuel loading which have produced historically uncommon large and severe stand replacing crown fires (Covington and Moore 1994; Allen et al. 2002). The effects of these fires include increased post-fire tree mortality (McHugh and Kolb 2003), initial decreases of understory plant cover (Springer et al. 2004; Hunter et al. 2006) and the subsequent susceptibility of the landscape to invasion of non-natives, soil erosion and flooding (Beyers 2004; Keeley 2004; Hunter et al. 2006).

On June 8th, 2006, a lightning strike ignited the Warm Fire on the northeastern edge of the Kaibab Plateau. The Kaibab National Forest had recognized the need to manage fire and fuels at a landscape-scale and developed a Wildland Fire Use (WFU) program with the Warm Fire being the first large implementation of that plan. The fire was initially managed as a WFU fire where it burned with a mix of low, moderate and high severity fire across approximately 7900 ha (19,500 acres) of predominately ponderosa pine forests (USDA USFS 2007). The initial plan was altered on June 25th, when weather conditions changed considerably, and the Warm Fire exceeded its maximum manageable area and was declared a wildland fire. The wildland fire portion of the fire was predominately high severity, but did burn in low and moderate severity and burned in all three vegetative types: mixed conifer forests, ponderosa pine forests and pinyon juniper woodlands. In total, the fire burned 24,000 ha (59,000 acres) with several very large high severity patches. The Warm Fire served as the first large application of the WFU program and had unintended consequences which resulted in controversial second-guessing of fire policies within northern Arizona and especially across the North Kaibab ranger district.

It is important to understand the ecological implications of such fire use and to apply that knowledge to post-fire rehabilitation activities such as seeding, as well as to the development of ecologically appropriate landscape approaches to fire management and forest restoration. The Kaibab Plateau served as a compelling location for linking fire effects analyses with landscape-scale approaches to post-fire rehabilitation because of its' isolation, high conservation value, and measurable legacies associated with historic wildlife, forests, and fire. For this research project we aimed to deepen our understanding and knowledge of the ecological impacts of fire and post-fire mitigation on the ponderosa pine forest community in northern Arizona.

Proposal objectives

Forest managers, stakeholders and researchers need to understand the costs and benefits of various fire management strategies and rehabilitation activities on the ecological integrity of the forests. The objectives for this project can be summarized under 3 major themes: (A) fire severity effects on the overstory composition and structure, understory vegetation and fuels; (B) post-fire rehabilitation effects on understory vegetation and; (C) landscape level fire effects and their relation to pre-fire characteristics. The key findings for each of these objectives is presented and the overall management implications for the study are discussed.

A. Fire severity effects on the overstory composition and structure, understory vegetation and fuels

1. *Determine correlation between fire severity, stand characteristics and tree mortality*
2. *Characterize the understory vegetation response to differing fire severities.*
3. *Assess fire effects on fuel load characteristics by fire severity*

B. Post-fire rehabilitation effects on understory vegetation

4. *Characterize the understory vegetation response to post-fire seeding.*

C. Landscape level fire effects and their relation to pre-fire characteristics.

5. *Refine fire severity maps for the Warm fire using Landsat imagery and field data validation one year post-fire.*
6. *Determine the correlation between measured fire severity and pre-fire predicted fire hazard characteristics.*

An additional objective originally listed in the proposal was to validate existing logistic regression models for post-fire mortality. We were unable to validate and build multivariable predictive models for ponderosa pine mortality because only 10 of 251 trees (4%) died by 2009, three years post-fire. Highly unbalanced data like this typically leads to overfit models with poor predicitive performance (Harrell et al. 2001). In the nearby Bridger-Knoll fire of 1996, McHugh and Kolb (2003) observed 13.9% mortality pointing to the unusual rate of mortality in our sites. However, only two of our 19 sites were located in areas of high fire severity and we had no moderate severity sites due to the experimental design for other objectives in this study. Trees that did die were from 10 cm dbh to 47 cm dbh showing no distinct trend in tree mortality.

Study design and methods

Our study was located in the Warm Fire on the Kaibab Plateau in the Kaibab National Forest in northern Arizona, USA (Fig. 1). Plot locations range in elevation from 2300 to 2590 m and at the landscape-scale have a similar disturbance history in terms of grazing and logging (Trudeau 2006). The fire burned across three vegetation communities: higher elevation mixed conifer (white fir (*Abies concolor* (Gordon and Glendinning) Hoopes), Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco) and blue spruce (*Picea pungens* Engelm.); mid-elevation ponderosa pine dominated interspersed with quaking aspen (*Populus tremuloides* Michx.), and Gambel oak (*Quercus gambelii* Nutt.) and lower elevation pinyon-juniper woodlands [*Pinus edulis* Engelm., *Juniperus osteosperma* Torr.) (USDA USFS 2007). This study was conducted in the ponderosa pine vegetation community. The understory was composed of common grasses, such as muttongrass (*Poa fendleriana* (Steud.) Vasey), squirreltail (*Elymus elymoides* (Raf.) Swezey), and Junegrass (*Koeleria macrantha* (Ledeb.) Schult) and common forbs including small leaf pussytoes (*Antennaria parvifolia* Nutt.), Fendler's sandwort (*Arenaria fendleri* A. Gray), and woolly cinquefoil (*Potentilla hippiana* Lehm.).

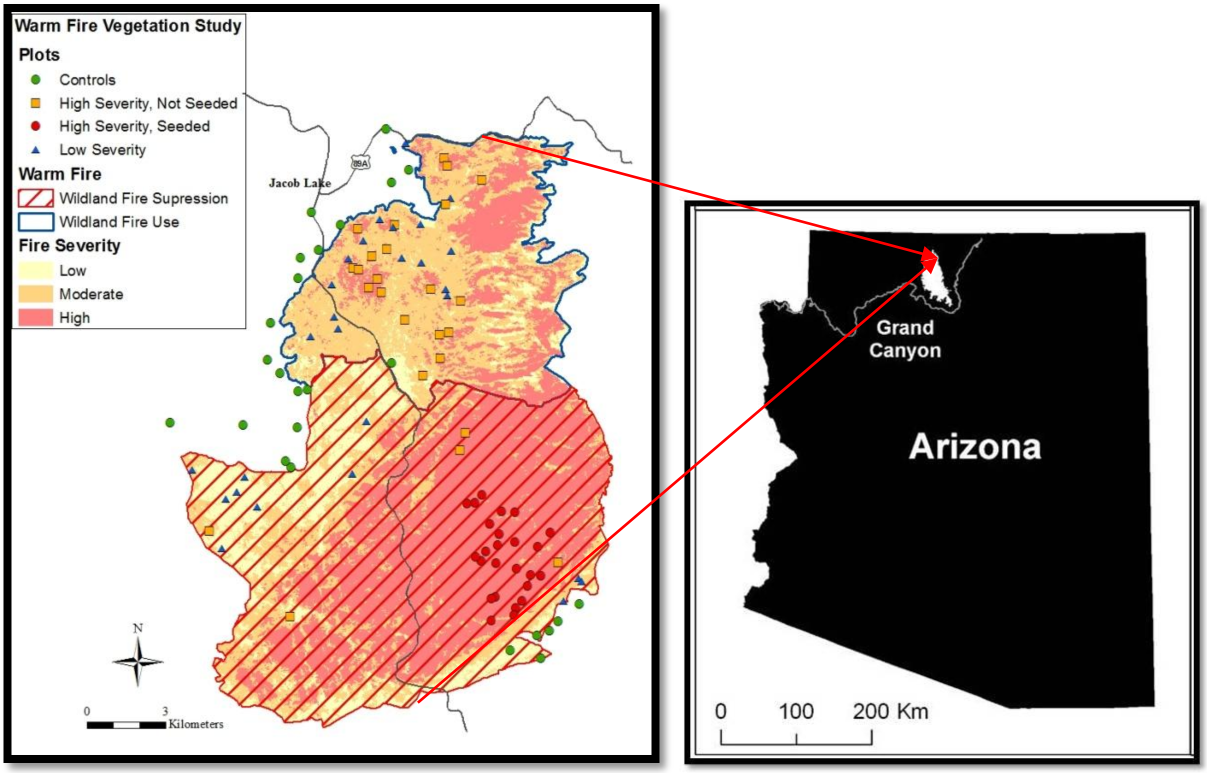

Figure 1. Location of study site. Perimeter of the Warm Fire; the northern portion of the fire (no crosshatch) was managed as a WFU while the southern portion (crosshatched) was managed as a wildfire and was subjected to post-fire mitigation. The fire encompassed 24,000 ha across three vegetation types. Burn severity is indicated by varying shades of grey. Unburned controls are indicated by green circles, low severity plots by blue triangles, high severity non-seeded by yellow squares, and high severity seeded plots by red circles.

The 14-year average annual precipitation was 61cm between 1995 and 2010. Roughly half of the precipitation comes during winter months in the form of snow while the remainder falls during summer monsoon rainstorms. Annual precipitation for the years covered by this study was 48 cm in 2007, 33 cm in 2008, 30 cm in 2009 (measured from October-September; Fig. 2.2). Precipitation in May and June of 2009 was unusually high whereas precipitation in July and August was lower than average. Temperatures range from an average January minimum of -5.7 ˚C and an average July maximum of 26.1˚ C (Western Regional Climate Center, 2009). Soils are derived from Kaibab Limestone parent material (Brewer et al. 1991).

Sampling design

Our study took advantage of a random disturbance (wildfire) and therefore we were unable to employ a more robust sampling design including replication and randomization (van Mantgem et al. 2001). Plots were stratified by fire severity, elevation, vegetation, slope, soils and BAER treatment. Sample points were randomly selected within strata using ArcView GIS software (ESRI 2006). High and low severity areas were delineated using the BAER burn severity map derived from Landsat satellite imagery and ground-truthed using the Composite Burn Index (Key and Benson 2006). Site elevation was restricted to within the ponderosa pine vegetation type on slopes less than 28 degrees. Soils were determined using Terrestrial

Ecosystem Survey of the North Kaibab National Forest layers and were restricted to Mollic Eutroboralfs (Brewer et al. 1991). Over 4,000 ha of moderate to high severity areas in all three vegetation types in the wildfire section were seeded with a non-native ryegrass (*Lolium perenne ssp.* multiflorum). Seeding was done immediately post-fire to reduce the risk of flooding, soil erosion and invasion of other more undesirable exotic species (USFS BAER 2006). Non-seeded sites were scattered across the fire in areas not subjected to post-fire rehabilitation (seeding) while seeded sites were restricted to the wildfire section (Fig. 1). The southern portion of the fire burned more consistently as a crown fire and was classified as a wildland fire receiving rehabilitation (suppression and seeding) treatments while the northern portion of the fire burned in a mosaic of fire severities. We controlled for environmental variation as much as possible, but the seeded sites are located in a different part of the forest; more consistently along exposed ridges when compared to non-seeded sites.

In 2007, as a pilot study one year post-fire, 42 total plots were permanently established and then measured for 2 additional years. There were 3 unburned controls, 10 low severity, 9 high severity non-seeded and 11 high severity seeded. We added plots to each treatment in 2008 and sampled a total of 102 plots in 2008 and 2009 for a total of 25 controls, 27 low severity, 25 high severity non-seeded and 25 high severity seeded. In 2010, 74 plots were sampled for overstory measures only as the area was opened to grazing. Unburned control plots were established within 1 km of the fire perimeter. Sampling was done during August and early September to capture the understory at greatest production. Sampling included overstory composition and structure, understory plant cover, species richness, and species composition, downed woody debris, and the composite burned index.

Field Sampling

Overstory and understory vegetation, fuels and burn severity were sampled throughout the Warm Fire. Sampling was based on a 12.6 m radius circular plot with two sub-plots and eight transects. A total of 13 frames for understory vegetation sampling were placed along the transects (Fig. 2). Topographic variables recorded at each site were slope (degrees), aspect, and elevation (m).

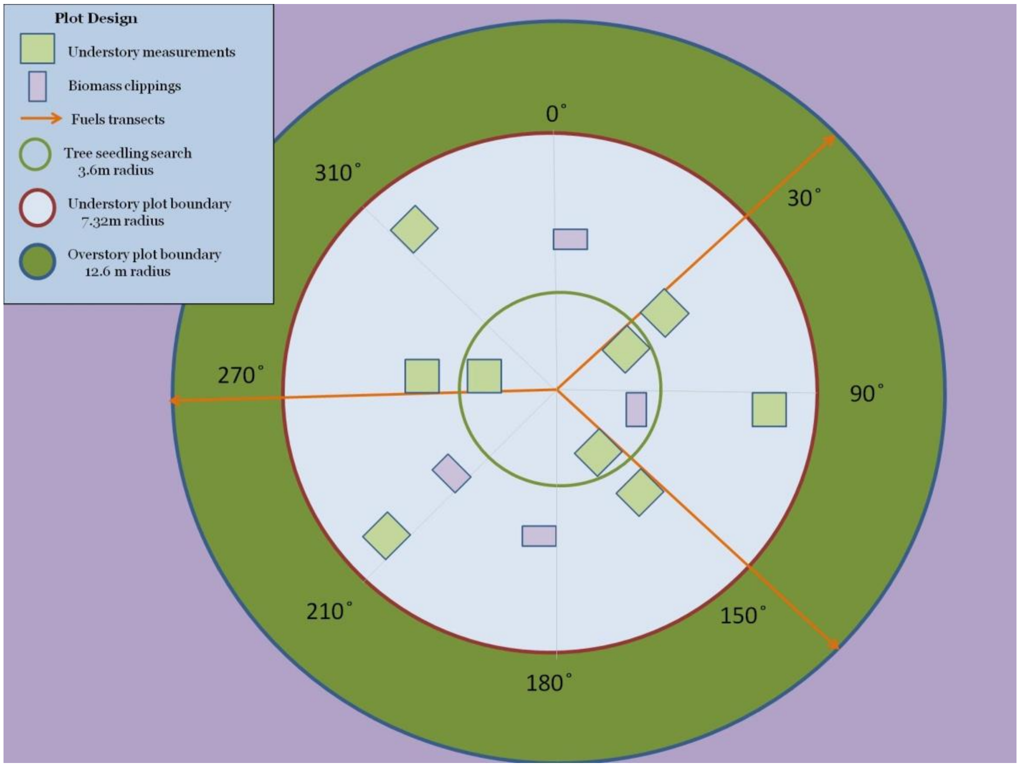

Figure 2. Plot layout. Overstory data was collected in the larger 12.6 m radius circle. A smaller sub-plot in the center (3.6 m radius) was used to count post-fire tree seedlings. Understory sampling was done in a 7.32 m radius circle with nine 1x1 m quadrats for vegetative and substrate cover and four 1x.25 m biomass frames. Species richness and shrub counts were collected from the entire understory circle. Fuels transects were 15 m long and located along three transect lines.

Overstory

We sampled overstory characteristics 74 plots across the Plateau. Within each plot, we recorded tree species, tree height, height to lowest live branch (crown base height), and DBH. For tagged ponderosa pine trees, tree crown and bole scorch sampling followed the methods outlined in McHugh and Kolb (2003). In general, the percent of the crown scorched and torched was assessed and bole char severity assessed using four classes: none, light char, medium char, and heavy char. Overstory canopy cover was measured with a densitometer in 2009 by counting the number of overstory hits at 33 points along the transect lines.

Understory

Understory vegetation sampling was done within the 7.32 m radius plot along six transect lines with a total of nine 1x1 m subplots (Fig. 2). Understory plant cover and foliar cover of vegetation are used interchangeably in this document. Foliar cover was estimated by botanists and calibrated for consistency in each quadrat by species, life form (graminoids, forbs) and total vegetation cover. On four .25 x 1 m frames, plants were clipped, separated by species, counted and then dried in an oven for 48 hours. Specimens were weighed and density recorded at the species level. Species richness was determined for the entire plot. Tree seedling density was recorded in a 3.6 m radius sub-plot. Species were classified into four life habits: annual/biennial graminoids (grasses and sedges), perennial graminoids, annual and biennial forbs, perennial forbs, shrubs and trees. Plant nomenclature and nativity are based on USDA-NRCS (2009) and voucher specimens are located in the Deaver Herbarium in Flagstaff, Arizona. We measured forest floor cover by estimating cover of bare soil, rock, wood, litter, duff, lichen, moss, and scat at each of the nine subplots.

Fuels

At 42 plots, we assessed fuel characteristics using a standard fuel sampling protocol following Brown et al (1974;1982). There were three 15 m transect lines along which we recorded 1- and 10- hour fuels in the first 2 m, 100-hour fuels in the first 4 m and 1000- hour fuels along the entire transect. Litter and duff depth were recorded starting at 2m and every 2m after to 14 m.

Composite Burn Index

The Composite Burn Index (CBI) (Key and Benson 2006) was used to rate fire severity on the ground at the plot level. The CBI plots were overlayed on the vegetation and fuel plots described above and a total of 71 CBI plots were collected in the summers of 2007 and 2008.

Statistical analyses for vegetation and fuels data

We analyzed all vegetation data using a permutational multivariate analysis of variance (PERMANOVA) (Anderson 2001; McCune and Mefford PC-ORD 5.1), which is a non-parametric test that can be used with non-normal univariate or multivariate datasets. We conducted one-way analysis using Bray-Curtis distance measures for multivariate data and Euclidean distance for univariate data using 9999 permutations, with significance at $\alpha = 0.05$. Differences in species richness were determined by the total number of species per plot per year in each treatment. Species that occurred in less than 5% of the plots were omitted from species composition analysis and ordinations but included in species richness and univariate analysis (McCune and Grace 2002). To specifically test if the seeded plots differed from the non-seeded plots in the rate of community change we used PC-ORD to calculate the Bray-Curtis distance for each plot. This method is similar to calculating the difference between two values in a paired t-test. This dissimilarity was analyzed with a PERMANOVA using Euclidean distance.

Non-metric multidimensional scaling (NMDS) was used to visualize the differences in plant community composition between seeded and non-seeded sites. Ordinations were done with PC-ORD V 5.1 (McCune and Mefford 1999) using Bray-Curtis distance measures with 250 runs with real data and 250 runs with randomizations, a maximum of 400 iterations per run and an instability criterion of .00001. Three dimensions were always recommended, however we chose the two axes that represented the most variation to create 2-D representations. We used Pearson's correlation coefficients to determine which species and covariates were most closely associated with the axes of the ordinations.

When differences in plant composition were identified in PERMANOVA analyses, we ran an indicator species analysis (ISA) to determine which species were driving those differences (McCune and Mefford 1999, PC-ORD V 5.1). ISA takes into account both relative abundance and relative frequency. Species with an indicator value > 30 and p > 0.05 were identified as indicator species (Dufrene and Legendre 1997).

We used PERMANOVA to test for the effect of fire severity level on fuel abundance. To test for differences in fuel abundance between years, we used a paired permutation test (Manly 1991). Under this resampling technique, fuel abundance values within a plot (each plot has two values, one for 2008 and another for 2009) are randomly assigned to “before” and “after” groups. Values are not shuffled between plots, thus preserving the temporal pairing of the study design. The mean difference between “before” and “after” groups is then calculated, and the process repeated many times. The proportion of resamples with mean differences that exceed the observed (i.e., actual) difference in fuel abundance between 2008 and 2009 is the p-value associated with the test. We used a similar approach to test for the significance of relationships between burn severity indices (i.e., the composite burn index and RdNBR, see next section below) and stand level overstory characteristics. In this analysis, however, values for the burn severity index were randomly permuted among plots because the study design was not paired. We also used the correlation coefficient as the test statistic. Permutation tests were performed in R statistical software (R Development Core Team 2010) using 9999 resamples.

Imagery Processing and Classification

Burn severity maps using Landsat 5 TM images of the Warm Fire area were created by the Monitoring Trends in Burn Severity (MTBS) program using the differenced Normalized Burn Ratio (dNBR) (Key and Benson 2006). The pre-fire Landsat image used for this analysis was taken on July 14, 2005, and the post-fire image was taken on July 4, 2007. All imagery pre-processing followed the MTBS protocols. Burn severity thresholds were assigned to classify the calculated dNBR values and create a map of fire severity for the Warm Fire (Fig.1). Thresholds for high and low severity were determined based on previous fires on the north rim of Grand Canyon National Park. The threshold for moderate severity was developed analytically using the following equation (Pabst pers. comm.):

$$\text{Moderate Threshold} = -28.811 + (0.372 * \text{Low Threshold}) + (0.539 * \text{High Threshold})$$

Because the dNBR method tends to underestimate high severity fire (Miller and Thode 2007, Soverel et al. 2010), the relativized dNBR (RdNBR) (Miller and Thode 2007) was also calculated and a corresponding severity map was developed (Fig. 2).

In a parallel attempt to estimate Warm Fire severity with respect to tree mortality, the Kaibab National Forest (KNF) used a Landsat Burned Area Reflectance Classification (BARC) map from the Forest Service Remote Sensing Applications Center (RSAC) (USDA USFS 2007) to classify tree mortality (USDA USFS 2007). The resultant GIS data layer was also used for assessment in this analysis as it was used by the Kaibab National Forest to assess post-fire management activities.

Evaluation imagery derived burn severity using CBI: correlations and accuracy assessments

Correlations and accuracy assessments between 71 CBI values (independent of the classifications described above) and the dNBR, RdNBR and tree mortality values were done to assess the validity of the imagery data. All spatial locations were plotted in ArcGIS 9.3 and projected to Universal Trans Mercator NAD 83 Zone 12 N to determine corresponding dNBR, RdNBR, and mortality values for a given CBI plot. Threshold values were then set in order to classify a plot within one of four severity categories. The CBI data could then be used to calculate Kappa statistics by developing error matrices for each classification scheme. Landis and Koch (1977) suggest three possible ranges for KHAT (the Kappa statistic): a value greater than 0.80 suggests strong agreement, a value between 0.40 and 0.80 suggests moderate agreement, and a value lest than 0.40 suggests weak agreement. Once the Kappa statistic has been calculated, a Z-test comparing the calculated Z statistic to the critical Z statistic (1.96 for α=0.05) can be used to determine whether a particular classification scheme performed better than another and whether that performance was better than could be expected by chance alone (Congalton and Green 1999). For the purpose of these analyses the dNBR and RdNBR classification of "increasing greenness" were re-classified as low severity. In addition, the "mixed-high" and "mixed-low" classification of the unmodified tree mortality scheme were re-classified as moderate severity.

Comparisons of modeled fire hazard to burn severity

FlamMap provides a number of output layers that can be used to assess fire hazard which can generally be defined as the types and amounts of fuels available to feed a fire (Sampson et al. 2000). FlamMap fire models for fire hazard and fire behavior were run using 97th percentile drought weather conditions, low fuel moisture conditions for live and dead understory fuels, drought condition foliar moisture, and 30 mph sustained winds from the SW (the direction of prevailing winds in the region; Lab of Landscape Ecology and Conservation Biology 2009a, b). These conditions are meant to represent the types of conditions that can result in extreme fire behavior. The model predictions were then compared to remotely-sensed values for fire severity based on the RdNBR analysis mentioned previously. Correlation was assessed by examining general patterns in descriptive statistics determined by using zonal statistics in ArcGIS 9.3. Zonal statistics summarize the values in the data raster layer based on the values in the zonal layer. For this analysis, the RdNBR classes of fire severity were used to define the zones for classification of the FlamMap predicted outputs for fire behavior and hazard. In addition, model outputs were assessed visually for large areas where predictions differed from RdNBR-assessed severity.

Key findings

A. Fire severity effects on the overstory composition and structure, understory vegetation and fuels

1. Determine correlation between fire severity, stand characteristics and tree mortality

A strong positive relationship existed in our sites between tree mortality and both RdNBR and CBI (Fig. 3). With respect to RdNBR, the relationship appears to be non-linear, where values greater than 400 correspond to high levels of mortality.

Relationships between tree density and basal area and both burn severity indices were negative, supporting the expectation that stands generally containing fewer but larger trees would have a lower fire severity rating (Figs. 4a-b and 5a-b). However, the relationships were only statistically significant for RdNBR. A significant positive relationship existed between height to live canopy and both RdNBR and CBI (Figs. 4c and 5c). However, this relationship can be slightly misleading in higher severity plots due to very few trees having a live crown. The "canopy height" could be a single "crown height". Relationships between tree size (dbh) and either burn severity index were not statistically significant (Figs. 4d and 5d).

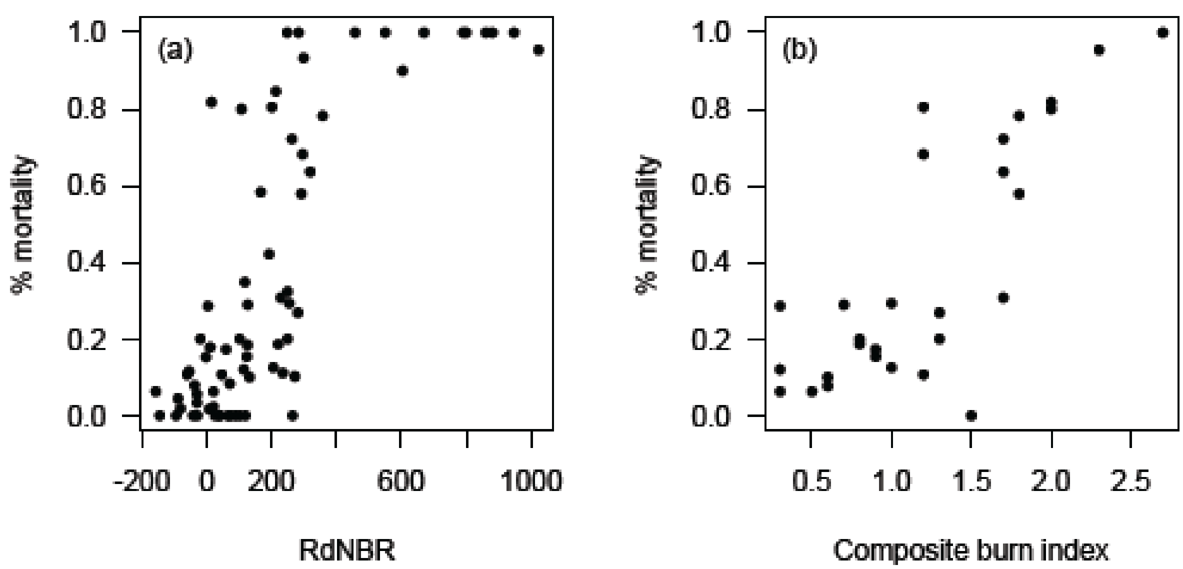

Figure 3. Plots showing the relationship between percent mortality (i.e., proportion of trees in a plot that were dead) and (a) RdNBR (r^2=0.78, p<0.0001) and (b) the composite burn index (r^2=0.80, p=<0.0001). The relationship between mortality and RdNBR appears to be non-linear.

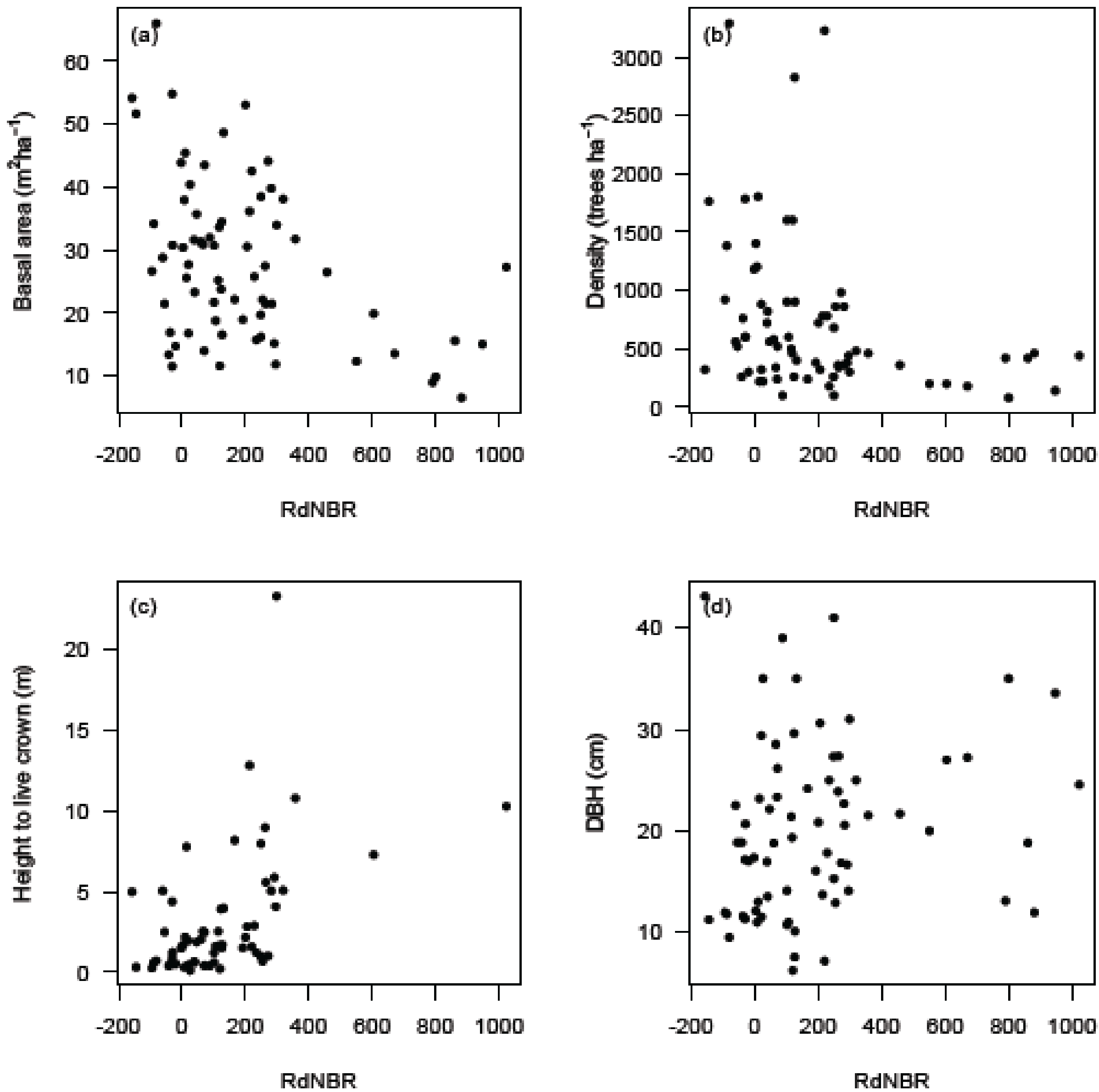

Figure 4. Plots showing the relationship between RdNBR and (a) basal area (r^2=-0.43, p<0.0001); (b) tree density (r^2=-0.32, p=0.008); (c) height to live crown (r^2=0.50, p=0.002); and (d) dbh (r^2=0.20, p=0.098).

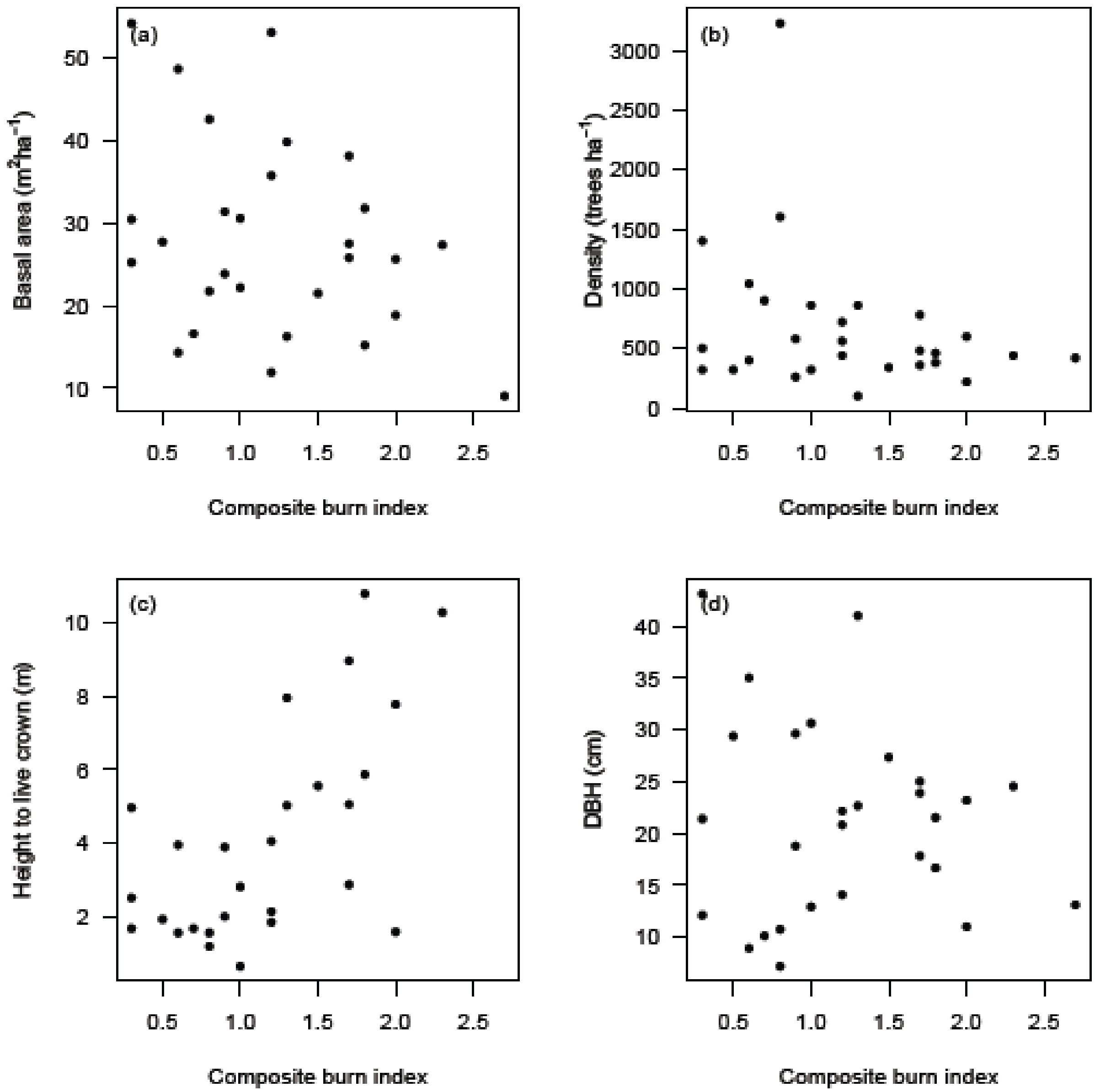

Figure 5. Plots showing the relationship between the composite burn index and (a) basal area (r^2=-0.33, p=0.089); (b) tree density (r^2=-0.29, p=0.129); (c) height to live crown (r^2=0.63, p<0.0001); and (d) dbh (r^2=-0.11, p=0.584).

2. Characterize the understory vegetation response to varying fire severities.

Fire in ponderosa pine forests stimulates the growth and diversity of the understory plant community. We observed this trend in our study as vegetation cover, biomass and species richness increased with increasing fire severity. High severity plots averaged the highest vegetation cover, biomass and greatest species richness with low severity plots a close second in species richness (Table 1). There was however, a lack of response in vegetative cover and biomass in the low severity plots. We attribute this to the unnatural accumulation of litter and duff from lack of fire in the last century. We saw that as litter cover increased, total vegetative cover and biomass decreased and conversely, when soil cover increased, so did vegetative cover and biomass (Table 1: Fig. 6). On low severity sites litter cover was still 53% two years post-fire while high severity sites had only 32% cover. Decreasing unnaturally high levels of litter through fire creates an opportunity for understory growth. Comparisons of pre-fire data collected on 16 of the plots indicate similar plant composition and structure to our unburned controls. Due to a small sample size we were unable to run statistical analysis, but Fig. 7 gives a good visual representation of pre-fire versus post-fire or unburned plots. When the community is separated into functional groups, we observed that the pre-fire cover of graminoids, forbs and shrubs is most similar to the unburned controls and low severity plots in 2009. Exotic species also appeared to represent about 10% of the total species in the pre-fire community (Fig. 7).

The plant community composition was significantly different between all three treatments; unburned, low severity and high severity sites (Fig. 8). The differences in the community were primarily driven by factors of fire severity: litter cover, canopy cover and exposed soil/rock (Fig. 8). Indicator species analysis showed significantly more annual and biennial forbs strongly associated with high severity and perennial forbs and trees as indicators of unburned controls and low severity. There were 23 species associated with high severity and of those 18 are annuals/biennials, 5 exotics, and 9 ruderals. Reference sites in ponderosa pine forests on the North Rim of Grand Canyon National Park that have not been subjected to fire suppression indicate that annual and biennial forbs are of great importance to the post-fire plant community for species richness and vegetative cover (Laughlin et al. 2004). Overall, we observed greater frequency and abundance of exotics as fire severity increased (unburned, low severity and high severity) and the total cover of exotics was less than 2% for all three treatments (Fig. 7). Prickly lettuce, yellow salsify, and dandelion were consistent in high severity plots but had low average cover (<1%). All three species are listed as noxious weeds in at least one state and are well adapted to disturbed areas as they are prolific seed producers with wind-dispersed seeds that can colonize from off-site. Cheatgrass and mullein were also more prevalent in high severity sites and cheatgrass in particular should be monitored to prevent spread across the landscape. Overall, we observed greater changes in understory vegetation, both positive and negative, within the high severity burn areas.

Table 1. Actual data values for all treatments in both years. Numbers represent foliar cover unless otherwise noted. Species richness is the average number of species/plot in the treatment. Letters indicate a significant difference ($p < 0.05$) among treatments in 2008 and 2009.

	2008			2009		
Category	U	L	H	U	L	H
Total foliar cover	7.535 a	8.627 a	21.68 b	8.021 a	8.924 a	23.26 b
Biomass	-	-	-	5.291 a	5.781 a	30.137 b
Richness	22 a	29 b	26 ab	25 a	30 b	32 b
Litter	73.71 a	53.61 b	32.95 c	79.52 a	69.20 b	45.96 c
Soil	3.731 a	11.27 b	20.36 c	1.614 a	6.494 b	9.045 b

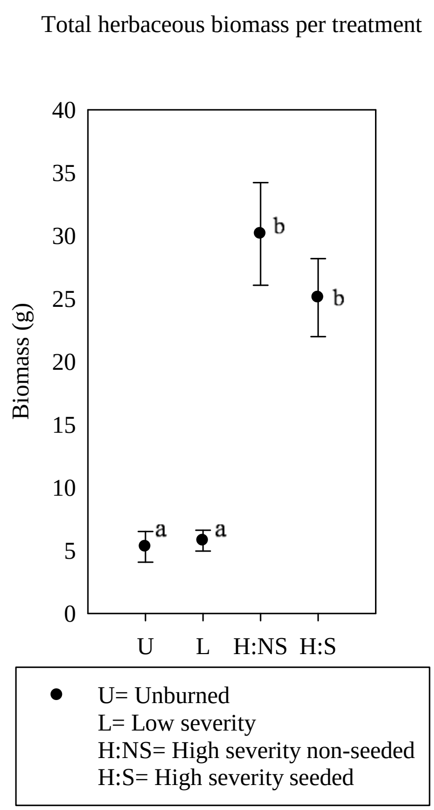

Figure 6. Average biomass in grams per treatment in 2009. For brevity, all treatments are included in one graph. Letters indicate a significant treatment difference at $p < 0.05$. Vertical bars represent +/- 1 standard error

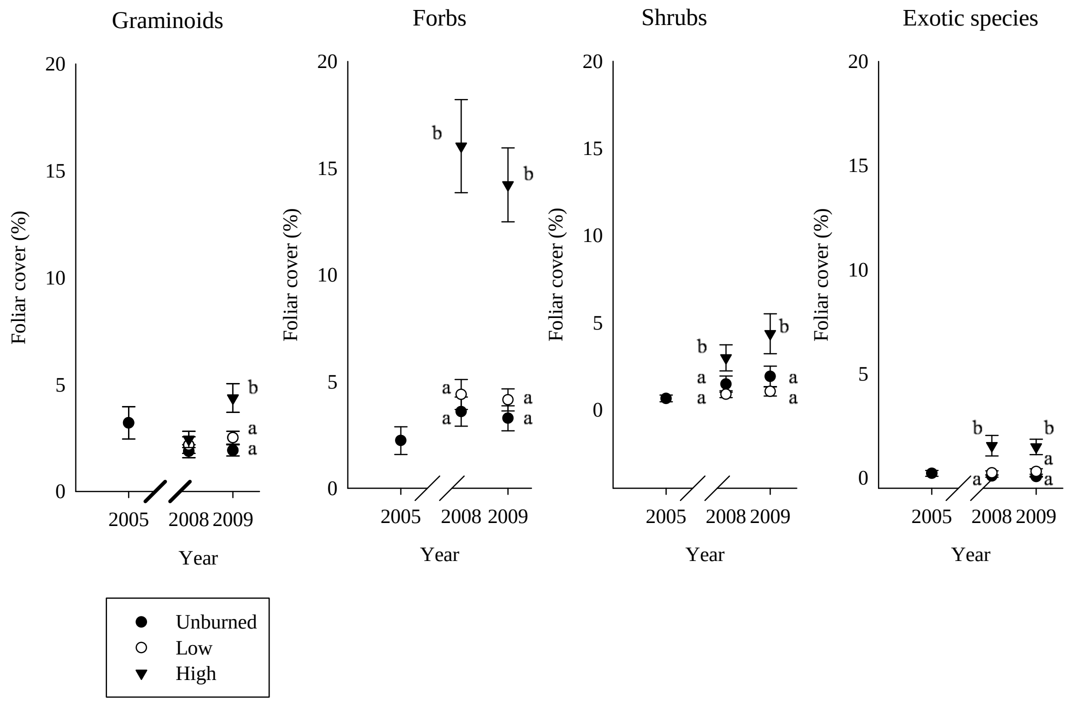

Figure 7. Comparison of total cover of graminoids, forbs, shrubs and exotic species from pre-fire data in 2005 compared to post fire data of 2008 and 2009. Data from 2005 is excluded from statistical analysis. Significant treatment effects ($p < 0.05$) are indicated with differing letters. Vertical bars represent +/- 1 standard error.

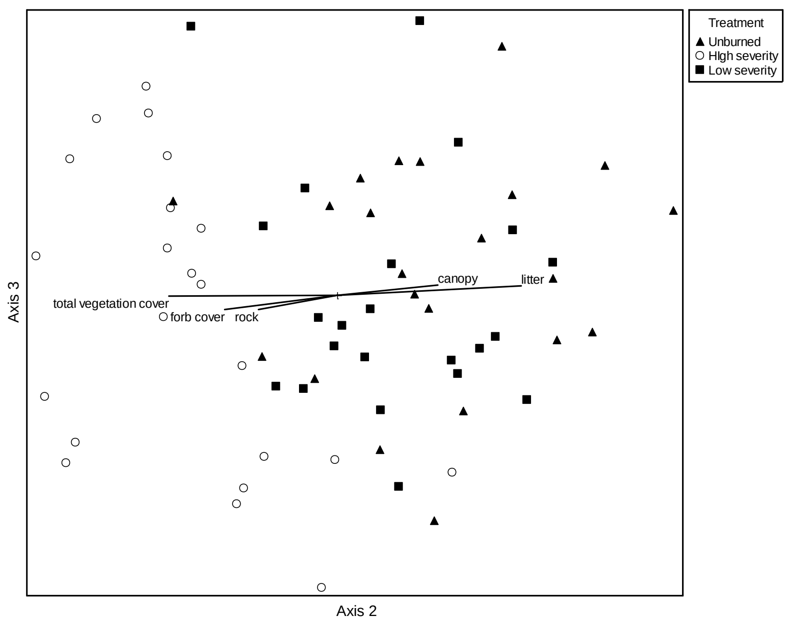

Figure 8. Non-metric multidimensional scaling (NMDS) ordination of plant community composition sampled across the Warm Fire ponderosa pine forests in northern Arizona in 2009 (three years post-fire). Pair-wise analysis using PermANOVA indicated that treatments were significantly different from one

another (U vs. L; p=0.025; L vs. H; p=0.0002; H vs. U; p=0.0002). The plot was constructed using 88 species found in 66 plots. The final solution had three dimensions (Stress= 17.69103, p< 0.01, 177 iterations).

3. Assess fire effects on fuel load characteristics by fire severity

High and low severity fire generally decreased the abundance of fuels on our sites at similar rates. For example, high and low severity sites contained significantly less total debris than unburned sites in 2008 and 2009 (Figs. 9d and 10d). These differences were largely driven by duff and litter, which on average composed > 87% of total debris (Figs. 9a and 10a). With respect to fine woody debris, high severity sites contained significantly less fuel than low severity and control sites in 2008 (Fig. 9b). By 2009, however, abundance of fine woody debris in low severity sites was more comparable to high severity sites than control sites (Fig. 10b). Low severity sites generally contained more coarse woody debris than high severity and control sites in both years, although differences among treatments were not statistically significant (Figs. 9c and 10c).

Mean abundance of fuels in the 1000-hr solid size class on high severity sites increased with time after fire and surpassed control sites by 2009 (Figs. Xc and Yc). Although this difference was not statistically significant, this trend might be perpetuated by increased downfall resulting from high severity fire. One might expect a similar trend with fine woody debris on low severity sites; however, this pattern is not evident from our data (Figs. Xb and Yb). On the contrary, fine woody debris decreased on low severity sites from 2008 to 2009. Data from four or more years post-fire may be necessary to observe an increase in FWD over time.

Our data does not support the expectation that fuel abundance should increase with time after fire (Fig. 11). In fact, abundance of duff and litter (and by extension total debris) in our sites was significantly less in 2009 than in 2008. In all other fuels categories, differences between 2008 and 2009 were insignificant. The decrease in duff and litter from 2009 to 2008 could be a combination of a heavy snowpack during the winter of 2008 and compaction of sampling sites due to sampling litter and duff in the same location both years.

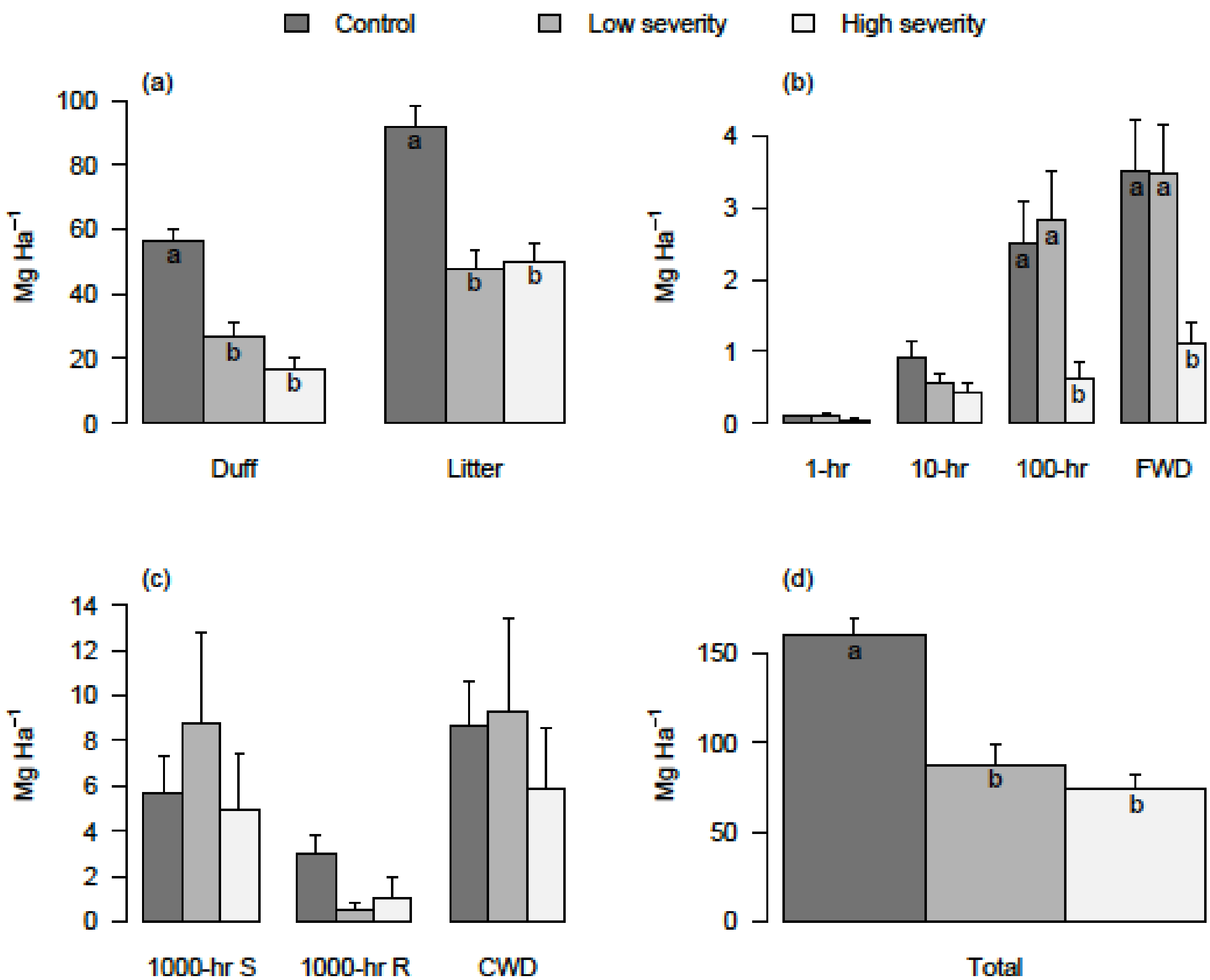

Figure 9. Abundance of fuels in sites during 2008. Bar plots show abundance of (a) duff and litter; (b) fine woody debris (FWD) including fuels in 1, 10, and 100-hr size classes; (c) course woody debris (CWD) including 1000-hr size class fuels classified as solid (S) and rotten (R); and (d) total debris. Standard error bars are displayed and letters show significant differences at $p < 0.05$. Absence of letters represents lack of significant differences.

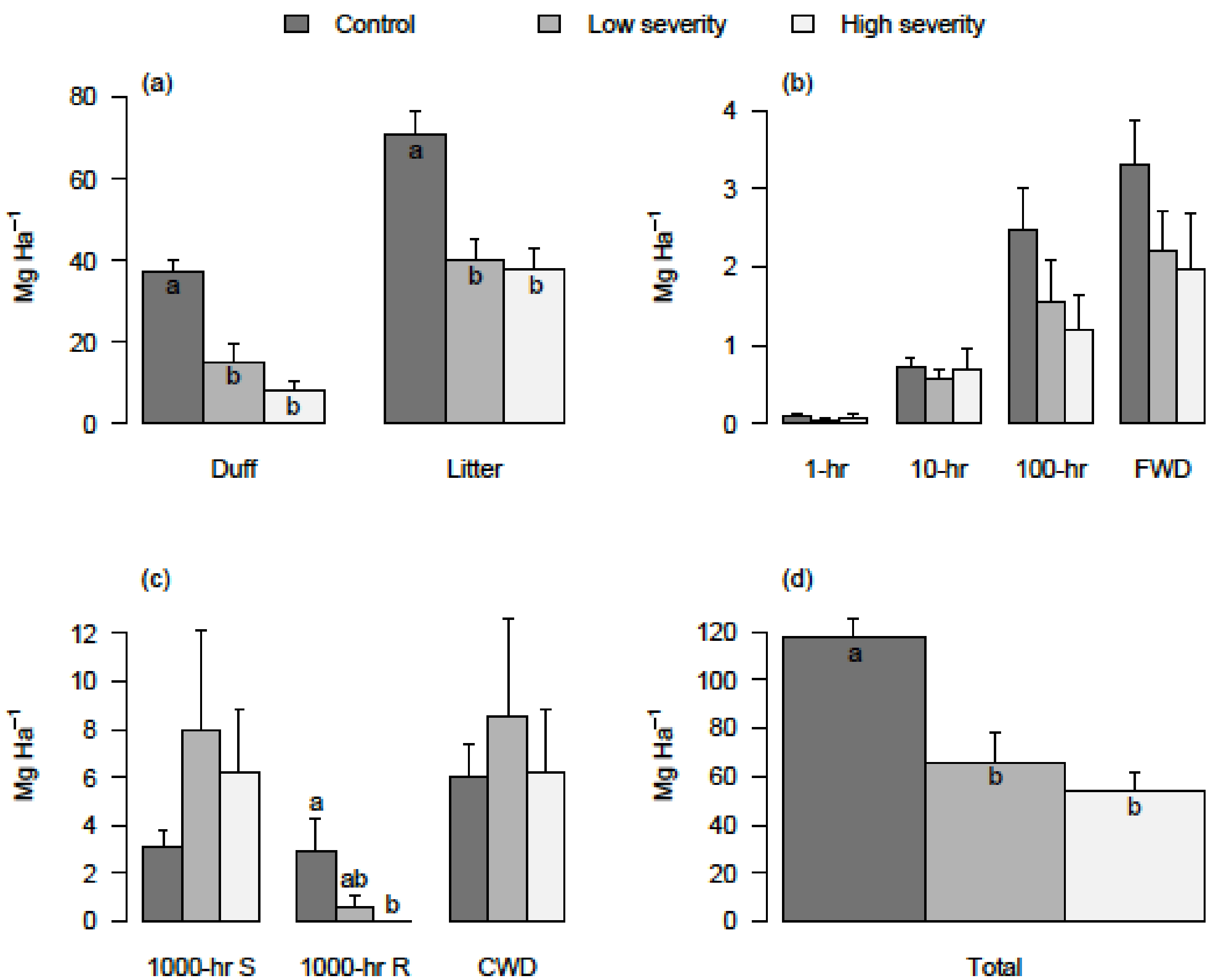

Figure 10. Abundance of fuels in sites during 2009. Bar plots show abundance of (a) duff and litter; (b) fine woody debris (FWD) including fuels in 1, 10, and 100-hr size classes; (c) course woody debris (CWD) including 1000-hr size class fuels classified as solid (S) and rotten (R); and (d) total debris. Standard error bars are displayed and letters show significant differences at $p < 0.05$. Absence of letters represents lack of significant differences.

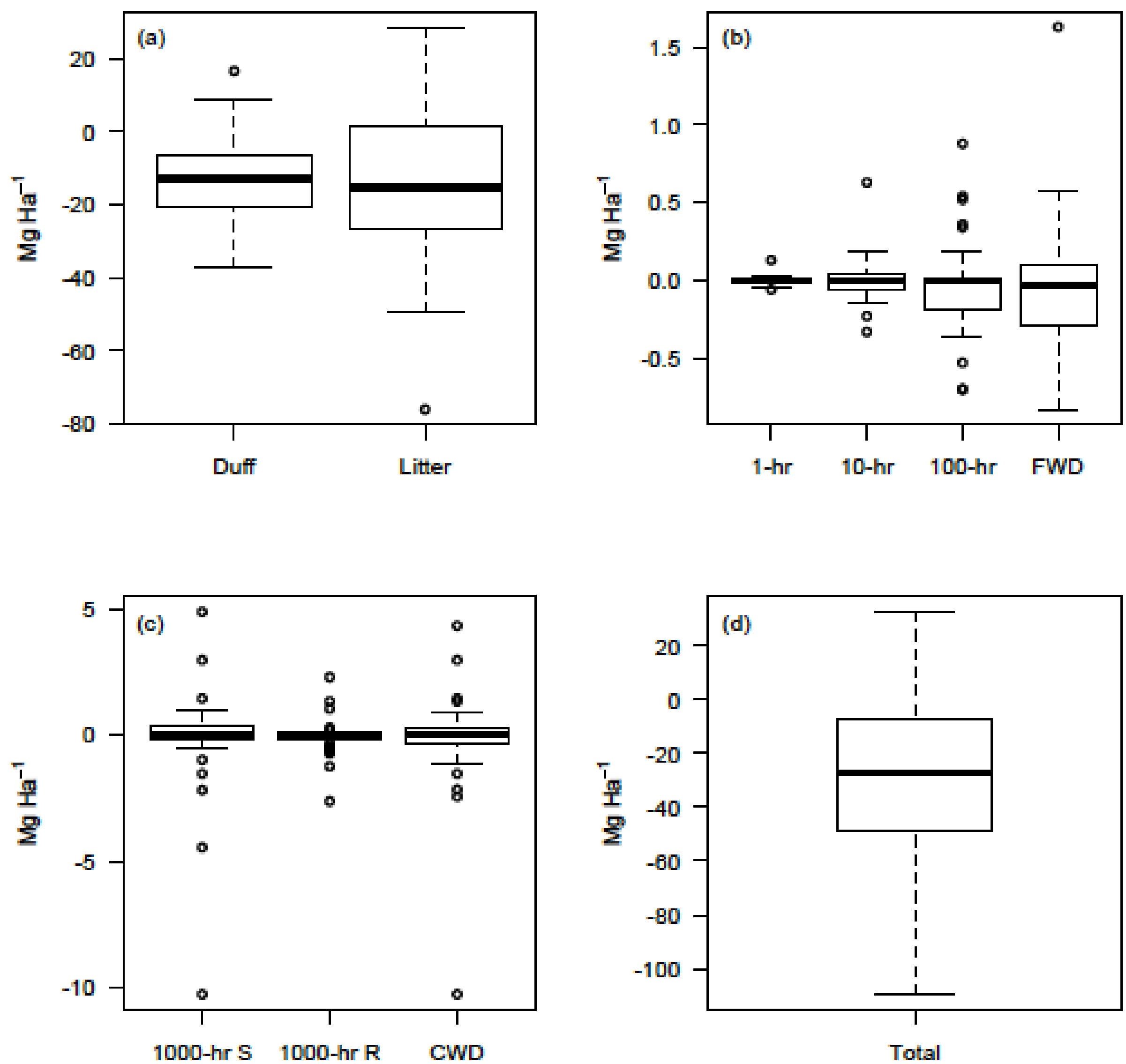

Figure 11. Difference in fuels abundance between 2009 and 2008 (i.e., 2009 fuels – 2008 fuels, such that negative values represent higher fuels abundance in 2008 than in 2009) over all fire severity levels. Box plots show differences for (a) duff and litter; (b) fine woody debris (FWD) including fuels in 1, 10, and 100-hr size classes; (c) course woody debris (CWD) including 1000-hr size class fuels classified as solid (S) and rotten (R); and (d) total debris. Tests of no difference between fuels in 2008 and 2009 were significant at $p < 0.05$ for duff, litter, and total debris only. For these categories, abundance of fuels at sites in 2008 was generally less than in 2009. The bold horizontal line in the box plot represents the median value. The bottom and top of the box show the 25th and 75th percentiles, respectively. The vertical dashed lines are the whiskers, which show one of two things. The upper whisker, for example, shows the maximum value or the 75th percentile plus 1.5 times the interquartile range (75th percentile – 25th percentile), whichever is smaller.

B. Post-fire rehabilitation effects on understory vegetation

4. Characterize understory vegetation response to post-fire seeding.

Total vegetation cover is used to determine the success of seeding and a recent review of published seeding studies concludes that while seeding is a common choice, it is not effective at producing the required 60% ground cover to reduce the amount of bare soil and prevent erosion and exotic invasions (Peppin et al. 2010). The results of this study support this review in that the seeded species averaged only 6% cover one-year post-fire and seed was most prevalent on gradual slopes. Seeded sites actually had less total cover and biomass than non-seeded sites in the Warm Fire although it was not a significant difference in either case (Fig. 6). Bare soil was also significantly higher in seeded sites hence seeding was not effective at decreasing the exposed bare soil (Fig. 12). Seeded areas also did not differ significantly from non-seeded sites in the cover of exotics species and the impacts of seeding appear to be relatively marginal on that front as well. The presence of cheatgrass in both seeded and non-seeded plots increased from 43% of all plots to 50% in 2009 with no significant difference in cover between treatments. Cheatgrass cover is low (< 2%) but the observed increase warrants concern and continued monitoring is strongly recommended. Ryegrass was still present across the landscape 3 years post-seeding and while the percent cover is decreasing, the presence of an exotic species occupies species space and usurps resources that would otherwise be available for native plant species.

The plant community composition was significantly different between seeded and non-seeded sites as confirmed by both statistical analysis and ordinations and thus the differences associated with seeding are disconcerting (Fig. 13). Although ryegrass cover was relatively low, there is evidence that it competes for space and may displace natives thus altering the initial community composition and possibly the community trajectory. The presence of ryegrass was associated with significantly less cover of three dominant native bunchgrasses (squirreltail, mountain muhly and muttongrass) and less cover of annual and biennial forbs in seeded plots. Annual and biennial forbs play an important role in post-fire restoration as they add significantly to the species richness and vegetative cover in burned areas in ponderosa pine forests on the Kaibab Plateau (Laughlin and Fulé 2008). We also observed significantly fewer ponderosa pine seedlings in seeded sites and acknowledge that this may be due to environmental variation as well as propagule pressure. We analyzed of the rate of plant community change in a treatment between years and results indicated that these communities are changing at similar rates, but potentially in different directions as per our indicator species analysis. Analysis of biomass data reflected similar trends with less biomass of native bunchgrasses and fewer annual and biennial forbs as indicator species in the seeded plots. The current effects of the ryegrass are not encouraging and the long-term persistence of this intentionally introduced ryegrass has the potential to further disrupt this system. Managers and stakeholders will need to continue monitoring to understand these long-term effects.

Table 2. A summary of total biomass (g), total vegetative cover (%) and exposed bare soil (%) in both seeded and non-seeded sites in 2009. Asterisks (*) indicate a significant difference ($p < 0.05$) between non-seeded and seeded plots.

	Biomass (g)	Vegetative Cover (%)	Bare Soil (%)
Non-seeded	30.137	21.59	9.05*
Seeded	25.080	19.03	19.08*

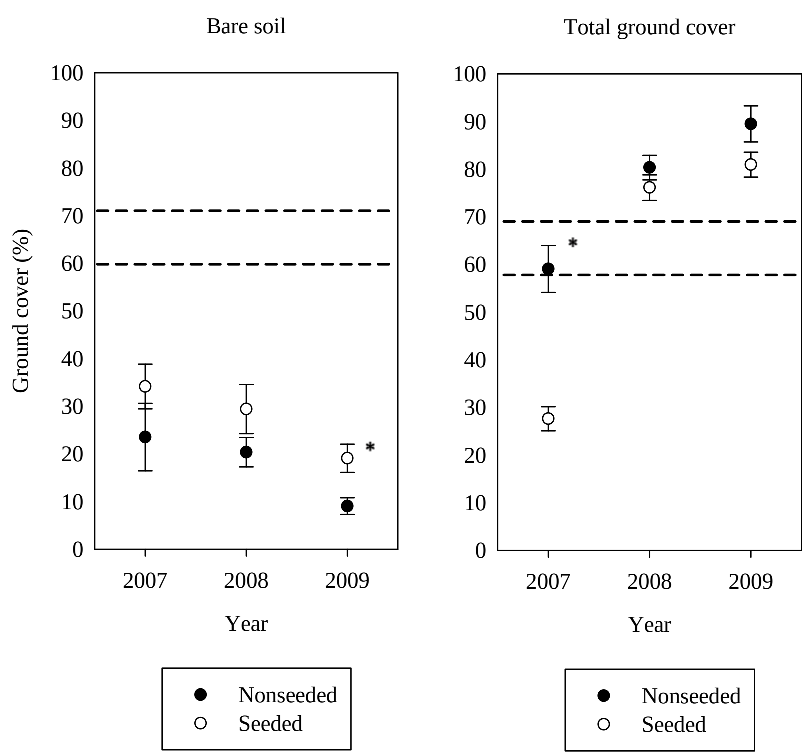

Figure 12. Average cover (%) of bare soil and all ground cover including litter, duff, rock, wood and vegetation in all three years in both treatments; the area between the dashed lines indicate the cover values at which erosion and runoff significantly increase for bare soil cover and decrease for total cover (Robichaud et al. 2000; Johansen et al. 2001). Asterisks (*) indicate significant difference at $p<0.05$. Vertical bars represent +/- 1 standard error.

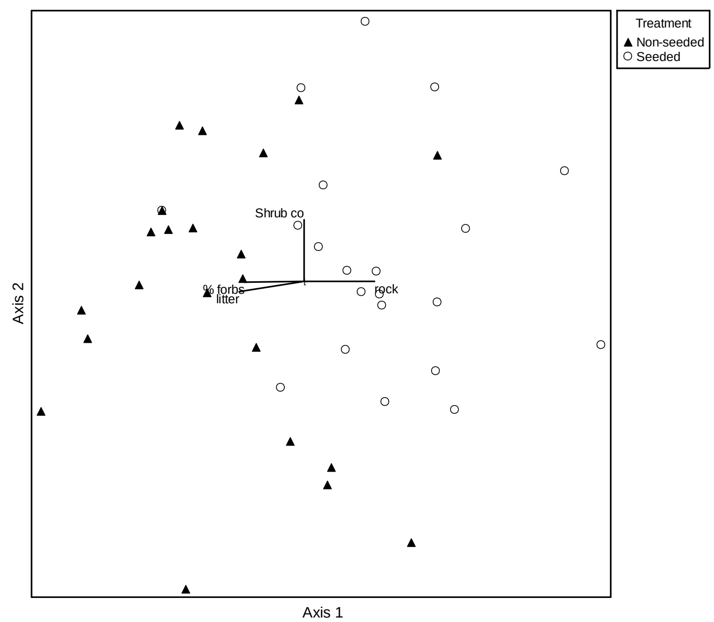

Figure 13. Non-metric multidimensional scaling (NMDS) ordination of plant community composition sampled across the Warm Fire ponderosa pine forests in northern Arizona in 2009. Permanova results indicated that treatments were significantly different from one another (p=0.0014). The plot was constructed using 109 species found in 44 plots. The final solution had three dimensions. (Stress= 17.46, p< 0.01).

C. Landscape level fire effects and their relation to pre-fire characteristics.

5. Refine fire severity maps for the Warm fire using Landsat imagery and field data validation one year post-fire

Estimates of the percentage of area burned within each severity class vary considerably depending upon the classification system used (Table 4: Fig. 14). The RdNBR model classified the greatest proportion of the landscape as having burned at high severity. In contrast, the dNBR model resulted in the lowest proportion of the area classified as high severity. However, it is worth noting that the tree mortality models, which are based on dNBR values with thresholds determined by CBI data, result in proportions that are similar to those resulting from the RdNBR model.

Remotely sensed values of fire severity were highly correlated with ground-measured CBI data, regardless of classification method (Table 5). Correlation coefficients were highest for RdNBR (0.84) and lowest for the modified tree mortality methods (0.75). Furthermore, evaluation of the Z-statistics for each image-classification method indicates that each method performed significantly better than could be expected by chance alone (Table 5).

Agreement between CBI data and the modified tree mortality map was the highest of all four methods considered (73.2 % overall accuracy) followed by RdNBR (59.2%) (Table 6). Similarly, KHAT values for the modified tree mortality and RdNBR methods indicate moderate agreement with CBI data. In addition, a Z-test comparing the two methods results in a Z-statistic of 1.645 indicating that the two techniques do not differ statistically from one another (Table 7). Furthermore, comparison between the dNBR and original tree mortality method resulted in a Z-statistic of 0.241 indicating that these two methods did not differ significantly. Finally, comparing the original tree mortality method with the modified method resulted in a Z-statistic of 2.17 indicating that the tree mortality maps significantly differed from one another.

Table 3. Thresholds used to determine burn severity class.

Severity Class	dNBR	r dNBR	CBI
No Data/Unburned	-970	20	0.49
Increased Greenness	-150	81	NA
Low	100	157	1.49
Moderate	218	249	2.49
High	390	250	3.0

Table 4. Percentage of burn area in each severity class for each method of imagery classification.

	Severity		
Method	Low	Moderate	High
dNBR	34.4%	34.6%	31.1%
RdNBR	35.5%	18.5%	46.1%
Tree Mortality (Original)	16.4%	38.0%	45.7%
Tree Mortality (Modified)	36.3%	17.9%	45.7%

Table 5. Correlation coefficients for comparison of imagery classification methods with ground-based CBI data. All p-values are < .00001.

Method	Correlation coefficient
dNBR	0.80
RdNBR	0.84
Tree Mortality (Original)	0.78
Tree Mortality (Modified)	0.75

Table 6. Agreement between the dNBR and RdNBR and CBI data (KHAT = Kappa statistic)

	dNBR		RdNBR	
Severity	Producer's Accuracy (%)	User's Accuracy (%)	Producer's Accuracy (%)	User's Accuracy (%)
Unburned	0.0	100.0	100.0	22.7
Low	43.3	61.9	50.0	60.0
Moderate	50.0	24.0	8.3	50.0
High	87.5	84.0	87.5	95.5
Overall	56.3		59.2	
KHAT	0.373		0.433	
Variance	0.0071		0.0066	
Z statistic	4.416		5.341	

Table 7. Agreement between the Kaibab National Forest tree mortality classifications and CBI data (KHAT = Kappa statistic)

	Tree Mortality (Orig.)		Tree Mortality (Mod.)	
Severity	Producer's Accuracy (%)	User's Accuracy (%)	Producer's Accuracy (%)	User's Accuracy (%)
Unburned	0.0	100.0	0.0	100.0
Low	16.7	41.7	76.7	69.7
Moderate	83.3	28.6	58.3	50.0
High	91.7	91.7	91.7	91.7
Overall	52.1		73.2	
KHAT	0.345		0.592	
Variance	0.0066		0.0064	
Z statistic	4.253		7.391	

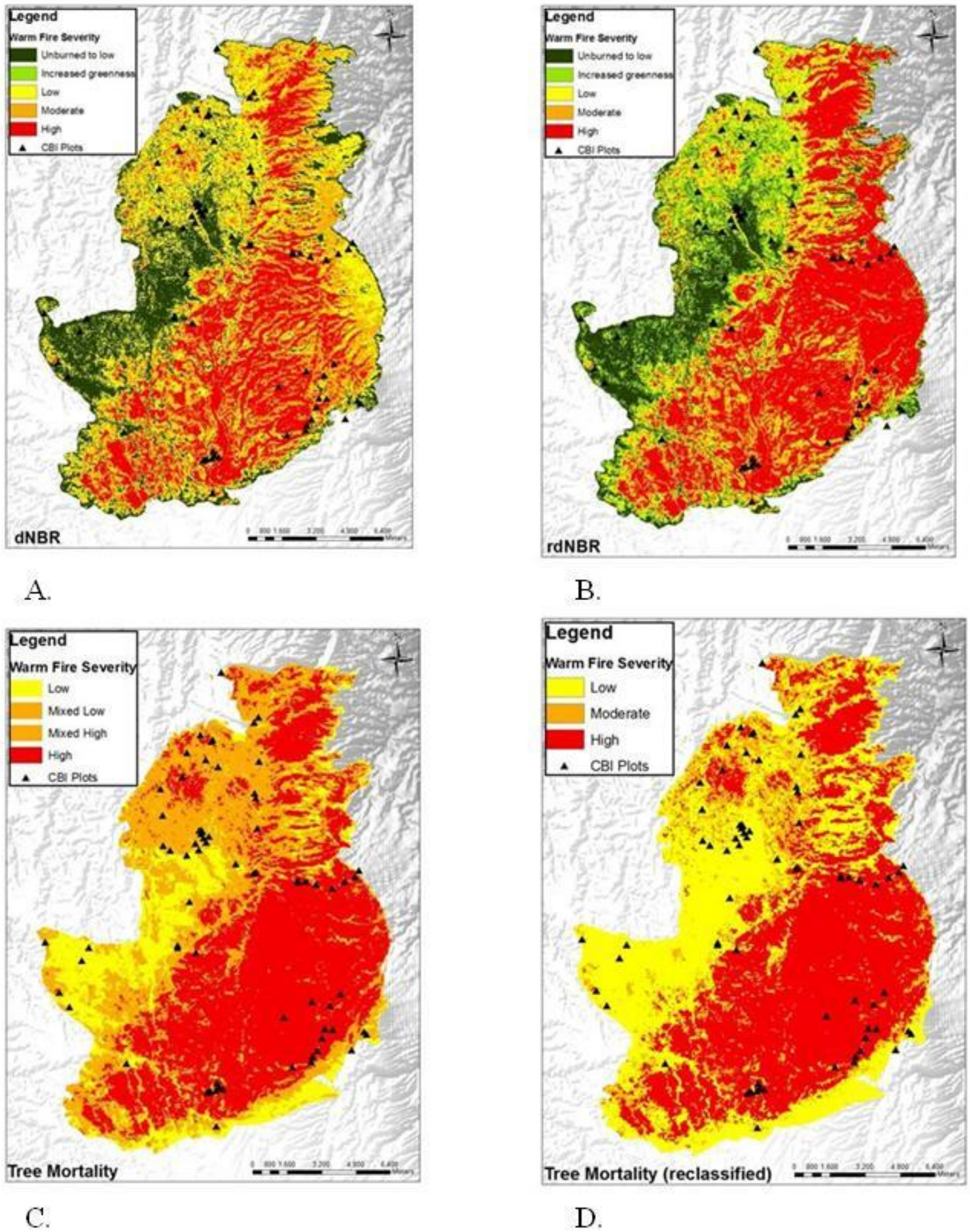

Figure 14. Burn severity maps based on the delta Normalized Burn Ratio (dNBR), and the relativised Normalized Burn Ratio (RdNBR). (A) Burn severity classification using the dNBR, (B) Burn severity classification using the RdNBR, (C) Tree Mortality classification of the dNBR from the Kaibab National Forest, and (D) reclassified tree mortality classification of the dNBR from the Kaibab National Forest

6. Determine the correlation between measured fire severity and pre-fire predicted fire hazard characteristics

Of the FlamMap outputs, crown fire behavior and heat per unit area probably provide the best representations of the relative danger presented by a fire as they depend primarily on the types and amounts of fuels available for a fire to burn. Higher values for FlamMap-generated heat and crown fire behavior layers depend on crown base height and crown bulk density. It should be noted that actual fire behavior is the result of numerous dynamic factors, not all of which can be modeled. FlamMap outputs are sensitive to values chosen for the various input parameters and uncertainty in input layers is likely to be passed on to the output layers. Additionally, the crown fire behavior layer does not give information about the likelihood of crown fire in a specific area, but provides information for how relative crown fire might occur over the landscape. As such, using the outputs as a generalization for comparison of locations on the landscape is more realistic than assuming the value for any pixel is "correct" (Lab of Landscape Ecology and Conservation Biology 2009a, b). Thus, attempting a correlation analysis with individual ground-based plots is likely inappropriate. However, comparing FlamMap predictions with modeled interpretations of severity at a landscape-scale does provide an opportunity to assess whether pre-fire predictions of patterns of fire hazard and behavior were similar to the patterns of burn severity as determined by the RdNBR (Miller and Thode 2007).

Summary zonal statistics indicate that patterns of remotely-sensed fire severity were similar to those seen in pre-fire predictions of fire behavior and fire hazard; however, substantial variability was present. Pre-fire predictions for fire hazard (expressed as kJ/m^2) generally increased with increasing severity. However, there was a considerable amount of variability within each RdNBR severity class (Table 8). Fire behavior model outputs are expressed as follows: 0 = no prediction, 1 = surface fire, 2 = passive crown fire and 3 = active crown fire. As such, the mode was used to assess the most frequently predicted fire behavior within each severity class. Surface fire was the most commonly predicted fire behavior within four of the five RdNBR severity classes. Predicted fire behavior was typically active crown fire in areas determined to have burned at high severity.

Visual comparison of model outputs with the RdNBR-generated severity map reveals two general areas where pre-fire predictions differed substantially from measured severity. These areas occur along the eastern flank and southwestern head of the fire (Fig. 15). Predictions for the western half were generally consistent with severity estimates with the exception of the aforementioned southwestern portion. These differences are likely the result of the fact that model predictions were based on winds out of the SW and the Warm Fire was driven primarily by a northerly wind (USDA USFS 2007). In addition, 97^{th} percentile weather conditions may not capture actual fire-weather conditions which may be more severe. Finally, the eastern portion of the fire burned into piñon-juniper woodlands which may have impacted fire behavior predictions due to dramatic differences in forest structure (B. Dickson, pers. comm.)

Table 8. Descriptive statistics for fire hazard (determined by predicted heat release in kJ/m2) and fire behavior (expressed as the mode where 0 = no prediction; 1 = surface fire; 2 = passive crown fire; 3 = active crown fire).

	Predicted Heat Release (kJ/m^2)				Predicted Fire Behavior
Severity	Minimum	Maximum	Mean	SD	Mode
Unburned to low	0.00	95325.20	15869.00	15410.80	1
Increased greenness	0.00	95325.20	16843.20	13776.30	1
Low	0.00	76053.80	17848.00	14028.00	1
Moderate	0.00	80332.10	18836.10	15166.70	1
High	0.00	91272.80	22708.90	17075.90	3

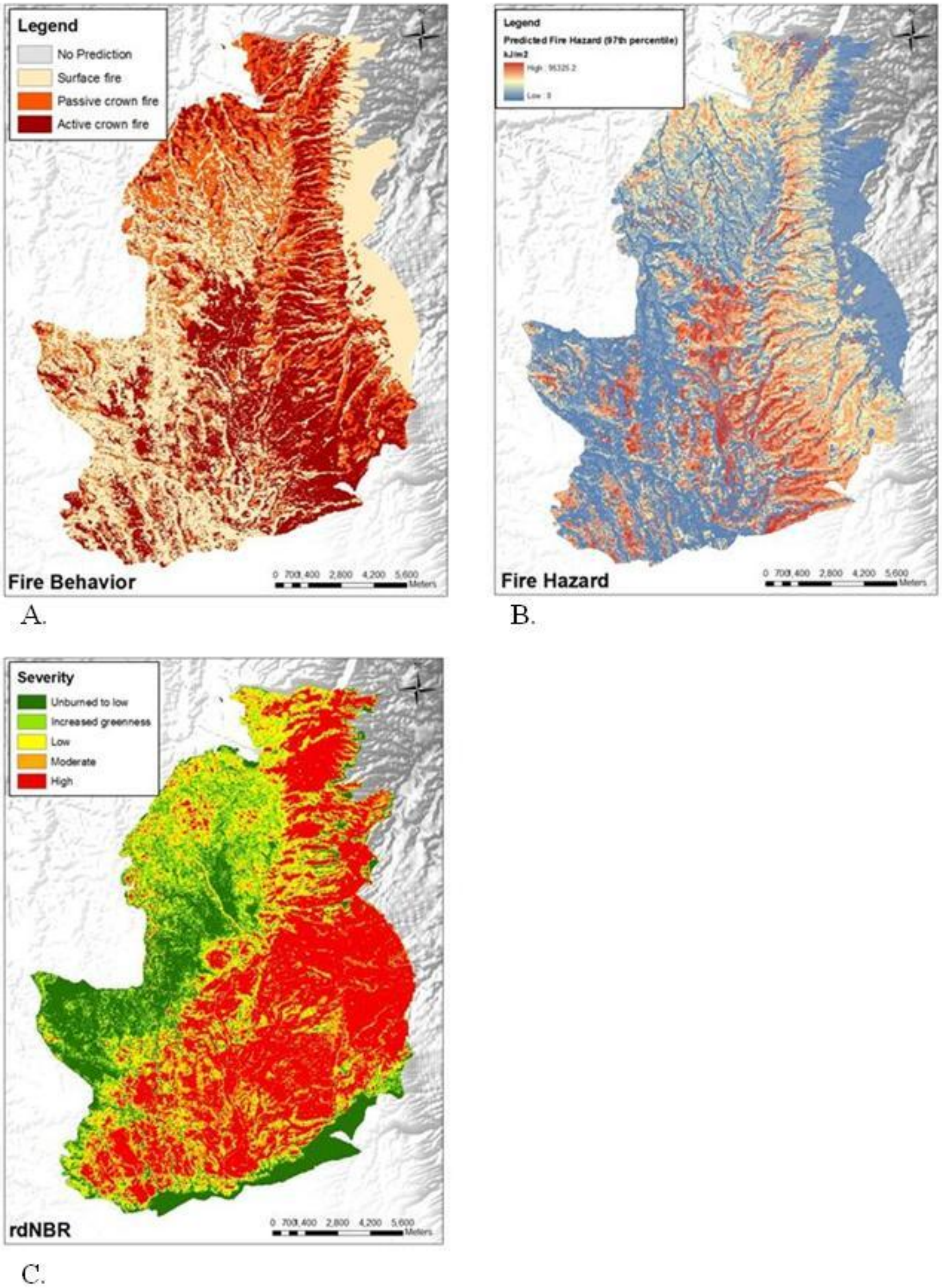

Figure15. Maps of (A) fire behavior, (B) fire hazard (expressed as kJ/m2), and (C) fire severity for the Warm fire. The fire behavior and fire hazard maps are the result of FlamMap modeling and the burn severity map was assessed using the relativise.

Management Implications

One major goal of the Wildland Fire Use program was to begin the process of reintroducing fire into ecosystems where historically it was prevalent and currently is lacking (USDA USNF 2007). New policies allowing wildfires to be managed in multiple ways for resource benefit will further the reintroduction of fire. With the Warm Fire, managers hoped to: decrease surface fuels; open the forest canopy to allow for the regeneration of pine trees and restoration of understory vegetation; reduce the threat of landscape-scale high severity fire; and return fire as an ecological process to the landscape. The WFU portion of this fire with the mixed burn severities should be viewed as a success in that it achieved most of the desired goals and moved closer to restoring the landscape. We observed an increase in annual and biennial forbs in both low and high severity burn plots but a positive response in vegetation cover only in high severity plots. However, it will be necessary to continue implementing fire across the landscape to continue the transition to an ecosystem with frequent fires; one low severity fire after 100 years of suppression is not enough of a disturbance to meet desired goals for forest composition and structure, understory vegetation and fuel loadings. The increase in exotic plant cover as a result of high severity fire warrants continued monitoring. Restoration projects such as landscape-scale thinning and targeted exotic removal may help to mitigate the increase in exotic species in the future. The major driver for changes in fuel loads between the controls, low and high severity was litter and duff fuel loadings. Overall, in both years the controls had close to twice as much fuel loading as both low and high severity areas. Continued monitoring of litter and duff loading will likely give managers an idea of when repeated treatments are needed to keep reduced fuel loads. Newly fallen trees will likely continue the trend of 1000-hr sound fuel loads increasing for low and high severity past control plot loadings.

Post-fire seeding has been shown to negatively affect plant communities and may contradict the ultimate goal of creating and maintaining ecologically stable and diverse ecosystems The conflict of seeding comes from the fact that success is determined by substantial cover of the seeded species to reduce bare soil and prevent exotic invasions but dominance of a single species has been shown to decrease the abundance of other, perhaps desirable species. The maintenance and rehabilitation of threatened watersheds is an important component of post-fire treatments, but seeding has not shown to be effective at increasing vegetation cover, decreasing bare soil and preventing exotic invasion. The results of this study add to that growing body of evidence that seeding is often not successful and may have unintended ecological consequences.

Comparison of four different remote-sensing based techniques for determining burn severity yields several conclusions. Estimates of the area burned within each severity class are highly sensitive to the technique used. This is of critical importance when rehabilitation decisions are based on remotely-sensed data. The tree mortality classification developed by the Kaibab National Forest resulted in severity estimates that were similar to those seen using RdNBR methods. In terms of expenditure of resources, it appears that using analytically developed RdNBR thresholds results in the greatest accuracy for the least cost. Results presented here indicate the necessity of using ground-based assessments of burn severity to develop better classifications and to assess the accuracy of remotely sensed data. Finally, the significant difference between the two tree mortality methods, where the data only differed in terms of how moderate-severity fire was characterized, indicates the sensitivity of these analyses to the thresholds that are chosen. Thresholds are somewhat subjective and the more field data available for certain types of vegetation, the better classifications will be.

The ability of FlamMap produced fire behavior and hazard predictions to depict general patterns of fire severity further emphasizes the utility of such tools for the purpose of planning at a landscape scales. However, high levels of variability and several areas where predictions were incorrect highlight two important considerations when using these model outputs. First, these models are meant to represent landscape-scale patterns and as such planning for specific pixels is inappropriate and likely to be inaccurate. Second, model outputs are dependent on the parameters used to generate the models. Thus, multiple iterations of the model may be necessary to truly understand the potential effects of fire within a given set of weather conditions. Furthermore, one can expect significant departure from predictions if the fire burns under conditions that differ substantially from those under which the model was run (e.g.: a substantially different wind direction). These considerations are of extreme importance to managers attempting to strategically locate fuels reduction treatments based on FlamMap outputs.

Relating to other research

- Peppin et al. 2010 conducted a systematic review of all studies done on post-fire seeding. Their results indicated that the highest quality studies showed that seeding was ineffective at providing adequate ground cover and did not provide enough cover to stabilize soils. They also reported that in half of the studies seeding was ineffective at curtailing exotic invasion and a majority of the studies reported that seeding with non-natives suppressed native plant regeneration.
- In 2008 Laughlin and Fulé demonstrated the importance of low severity fires in relic ponderosa pine forests in Grand Canyon National Park. These historic fires promote species richness, especially in terms of annual and biennial forbs and provide the ground cover to help stabilize soil after a fire.
- A current project with Grand Canyon National Park is looking at tree regeneration and fuel changes relative to time since the last fire, burn severity and the number of times burned for the transition zone between mixed conifer and ponderosa pine.

Future Research Needs

- Long-term studies are greatly needed to determine the ultimate effects of high severity fire and post-fire seeding on the floristic composition. Plots installed for this study should continue to be sampled to ascertain the long term effects and trajectory of the understory and overstory communities.
- There is also a lack of information of specific species response to fire and seeding. While the Fire Effects Information System developed by the Rocky Mountain Research Station provides a clearinghouse for species response to fire, there are very few species with complete information (USDA FEIS 2009).
- Additional studies on ponderosa pine regeneration following fire and seeding would be beneficial for managers when determining if seeding is the appropriate measure for post-fire rehabilitation.

- Data from this project can be used to model future conditions using the Forest Vegetation Simulator with the Fire and Fuels Extension. Different management scenarios could be evaluated in their ability to meet desired conditions. This could include information on how often stands need to be burned and what mechanical treatments may be needed. In addition, the effects of climate change could be addressed for these stands and future fire behavior.

Deliverables Cross-Walk

Proposed	Delivered	Status
M.S. Thesis	McMaster, Melissa Anne. Effects of fire and post-fire seeding on understory vegetation in a ponderosa pine forest in northern Arizona. August 2010.	Completed
Journal Articles	McMaster, Melissa A., Andrea Thode, Michael Kearsley. Effects of post-fire seeding with ryegrass (*Lolium perenne spp. multiflorum)* on understory plant communities. *In prep*	To be submitted to the *International Journal of Wildland Fire*
	McMaster, Melissa A. Andrea Thode, Michael Kearsley. Understory vegetation response to varying fire severities in a ponderosa pine forest in Northern Arizona. Anticipated date of completition: December 2010	To be submitted to the *Journal of Vegetation Science*
Field Trips	Native plant regeneration and understory plant community response following post-fire rehabilitation seeding in the Warm Fire. Presented at the "Colorado Plateau Chapter of the Society for Conservation Biology: Annual meeting". Warm Fire Field Trip. October, 2008.	Completed
	A summary of results for the Warm Fire Study. Kaibab National Forest, Grand Canyon Trust and Northern Arizona University participants	Scheduled for October 2010
Presentation at the Kaibab National Forest	Changes in Vegetation and Fuels on the Warm Fire, Kaibab Leadership Meeting, Williams, AZ. April, 2009.	Completed
	Effects of fire and post-fire seeding on understory vegetation in a ponderosa pine forest in northern Arizona, Kaibab National Forest, Williams, AZ. June, 2010.	Completed
Presentation at Scientific Conference	Seeds of change: A comparison of seeding vs. natural recovery for post-fire rehabilitation in a ponderosa pine forest. 2010. 95th Annual Meeting, Ecological Society of America, Pittsburgh, PA. August 2-8, 2010.	Completed
	Effects on native plant regeneration and understory community response three years post-fire and after seeding with Lolium multiflorum in a ponderosa pine forest in northern Arizona. 2009. Association for Fire Ecology – 4th International Fire Ecology and Management Congress, Savannah, Georgia. November 30 – December 4, 2009.	Completed
	Effects on native plant regeneration and understory community response following a post-fire seeding of Lolium multiflorum in a ponderosa pine forest in northern Arizona. Poster. Wildfires and Invasive Plants in American Deserts Conference and Workshop. Reno, Nevada.. December 9-11, 2008.	Completed
	The Warm Fire's effects on understory vegetation, ponderosa pine mortality and fuels: Implications for post-fire management. Poster presentation at "Fire in the Southwest: Integrating Fire into Management of Changing Ecosystems." January 2007.	Completed
Progress Report to the Kaibab National Forest	Changes in Vegetation and Fuels on the Warm Fire, Kaibab Leadership Meeting, William, AZ. April, 2009.	Completed
	Effects of fire and post-fire seeding on understory vegetation in a ponderosa pine forest in northern Arizona, Kaibab National Forest, Williams, AZ. June, 2010	Completed
Project Info on Websites	www.grandcanyontrust.org; http://www.for.nau.edu/cms/content/view/885/1395/	Update as needed

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