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For the Liberal Arts, the Environment, and Social Responsibility

# **Project Title**: Effects of fires and insects on fuel structures in piñon-juniper and post-fire invasive communities

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**Principal Investigator:**

Dr. Lisa Floyd-Hanna, Prescott College, 220 Grove Ave. Prescott, AZ , 86301,
lfloyd-hanna@prescott.edu

**Co-Principal Investigators:**

Dr. Tim Crews, Prescott College, Prescott, AZ
Dr. William H. Romme, Colorado State University, Fort Collins, CO

**Collaborators:**

David Hanna, Prescott College, Prescott, AZ
Dr. Monique Rocca, Colorado State University, Fort Collins, CO
Dustin Hanna, University of Arizona, Tuscon, AZ
George San Miguel, Mesa Verde National Park. CO

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## ABSTRACT

In the last decade, piñon-juniper (*Pinus edulis-Juniperus osteosperma)* woodlands of southwestern Colorado have been heavily impacted by wildfires and insects, and the effects on fuel structures and future fire intervals have been unclear. In these piñon-juniper woodlands, pre-historic fire rotations of up to 400 years were documented, but in recent climatic conditions, large stand-replacing wildfires burned between 1989 and 2008. Since the 1990s, a period of drought and rising temperatures, portions of the post-fire landscape were targets for invasive *Carduus nutans*, muskthistle and *Bromus tectorum*, cheatgrass, whose presence create unprecedented fuels, may shorten fire intervals, and reduce native biodiversity. We predicted that extensive tree mortality would alter fuel properties and fire behavior in dense unburned woodlands. In his two-year study, we addressed changes in fuel structures due to recent fires and insect infestations and explored management implications of these changes in Mesa Verde National Park and adjacent portions of the Ute Mountain Ute Tribal land.

Given the widespread nature of the mortality event of 2002-2004 we found few unaltered woodlands to serve as undisturbed controls. Therefore, also we re-evaluated nine fuel profiles originally sampled in 1993-1995 to provide pre-mortality fuel conditions. Woodland canopy and horizontal continuity of crown fuel has been significantly reduced. We found an increase in litter, 1to 100 hr fuels, and 1000 hr fuels following the mortality event. Woodland structural diversity as has decreased. However, the structure of post-mortality woodlands remains within the historic range of variability (HRV) for this region. Fire behavior has not changed significantly except for increased heat release.

In contrast, fuel structures and predicted fire behavior vary significantly across a time-since-fire chronosequence. Higher spread rate and intensity values are modeled in mid-successional areas, but measures decline as piñon-juniper woodlands get older and are lowest in old-growth stands. Early successional stages (i.e. burns in last 25 years) support *Bromus tectorum*, cheatgrass, which presents unprecedented levels of continuous surface fuel. Post-fire mitigation sponsored by the Federal BAER program using native perennial grasses in 1996 and 2000 significantly reduced the abundance and cover of invasive plant species which will likely have a significant effect on frequency of future fires. Native species diversity and abundance were not reduced by post-fire mitigation, nor do we detect any changes in fire behavior due to the introduction of perennial grasses. Management implications of these trends are discussed in the final section of this report.

## BACKGROUND AND PURPOSE

Piñon-juniper (*Pinus edulis-Juniperus osteosperma)* woodlands of southwestern Colorado and the adjacent Four-Corners states have been heavily impacted by wildfires, drought, and insects during the past decade, and projections indicate that these kinds of disturbances will continue or even increase in coming decades over much of the southwestern U.S. Disturbances of this frequency and intensity were not experienced during the 20th century. Prehistoric fire intervals in southwestern Colorado's piñon-juniper woodlands were centuries long (Baker and Shinneman 2004; Floyd et al. 2000, 2004; Shinneman and Baker 2009), yet under recent climatic conditions, mature woodlands have burned frequently and extensively, with more than half of the area of Mesa Verde National Park (MVNP) burned in the last two decades (Floyd et al. 2004). In addition to fire disturbance, piñons in the Four-Corners region have been subjected to intensive *Ips confuses*, pinon engraver bark beetle, infestations. Juniper mortality also has occurred, but with lower severity and often unknown causes. Overall, more than one third of the conifers in MVNP and neighboring Ute Mountain Ute (UMU) tribal lands have died in the past decade (Breshears et al. 2005, Floyd et al. 2009).

Recovery from such disturbances now must occur under unprecedented warm and dry conditions. Today, portions of the post-fire landscape are targets for invasive plants, notably *Bromus tectorum*, cheatgrass, whose presence may create continuous fuel conditions and may further shorten future fire intervals, especially if drought and rising temperatures continue (Belnap et al. 2003, Billings 1990, Brooks et al. 2004, D'Antonio and Vitousek 1992, Link et al. 2006, Miller et al. 2001, Whisenant 1990). Post-fire mitigation funded by the Federal Burned Area Rehabilitation Program (BAER) has been conducted on Mesa Verde by means of aerial seeding of native grasses (Floyd et al. 2001), but the long-term effectiveness of this treatment in suppressing non-natives and facilitating a natural post-fire successional trajectory is not yet known. The effects of these intense disturbances on fuel structures and future fire frequency and severity, and the effectiveness of post-fire mitigation treatments throughout Southwestern woodlands, are unclear; hence the need for this study.

In this 2-yr project, we conducted two related field-based studies to evaluate how the piñon-juniper vegetation and fuel structures in MVNP and UMU have changed as a result of recent fires and insect infestations. We also investigated the influence of invasive plant species and post-fire mitigation treatments on current and projected vegetation structure and fuels conditions. We further evaluated the implications of measured changes in fuels structures using fire behavior models (NEXUS) to simulate rate of spread, intensity, and potential for torching and crowning.

The specific questions that we addressed in this research were as follows:

**Question 1**: ***How have fuel profiles, potential fire behavior, and woodland structure in unburned piñon-juniper woodlands changed since the 2002-2004 mortality event?***

**Question 2**: ***How do fuel profiles, potential fire behavior, plant diversity and structure differ between recently burned areas and across the fire chronosequence?***

**<u>Question 3</u>:** ***How does community composition differ between recently burned areas that were or were not treated with post-fire seed applications, with an emphasis on comparing abundance and richness of native vs. non-native species?***

## STUDY DESCRIPTION AND LOCATION

We conducted field sampling on the Mesa Verde cuesta, a prominent landform in southwestern Colorado which includes MVNP on its northern portion and the adjacent UMU on the southern portion. The cuesta is covered by a mosaic of old-growth piñon-juniper woodlands, some of which have not burned for hundreds of years (Floyd 2003), interspersed with post-fire successional communities resulting from fires in 1934, 1959, 1972, 1989, 1996, 2000, 2002, 2003, 2005 and 2008. Erdman (1970) described a typical post-fire successional trajectory in piñon-juniper woodlands of MVNP, in which 300+ years were required to restore the original old-growth structure and composition. However, the extent and severity of recent fires, coupled with the invasion of burned areas by non-native plants and the potential for more frequent burning in the future, may have set the stage for novel successional pathways, vegetation structures, and community composition and biodiversity .

The current study builds in part upon a previous Joint Fire Science Program JFSP project that we completed in 2004. In that previous study, our research team investigated patterns of weed (primarily *Carduus nutans*, muskthistle, and *Cirsium arvense*, Canada thistle) distribution in MVNP to identify plant communities most vulnerable to post-fire weed invasion, and we created a spatially explicit "weed-risk" model to predict the most vulnerable sites (Floyd et al. 2006). This study demonstrated that at the scale of the entire park, mature piñon-juniper woodlands growing on two particular soil types were most vulnerable to post-fire weed invasion. The study also revealed that mountain shrublands on all soils were the least vulnerable to invasion because these communities are dominated by native species capable of rapid and vigorous re-sprouting after fire. At a finer scale, greater richness of native species was correlated with greater numbers of non-native species, indicating that habitats with high native biodiversity are at the greatest risk of weed invasion. In unburned areas, muskthistle density increased with increased soil nitrogen, phosphorus, and salinity; and in burned areas musk thistle increased with soil nitrogen and silty soil texture. We also assessed the effectiveness of aerial seeding with native grasses conducted within a few weeks after fire (referred to as "immediate seeding" in the paragraph below); these treatments were funded by the Burn Area Emergency Rehabilitation Program (BAER) in 1996 and 2000, and proved to be the most effective treatments in reducing post-fire weed densities (BAER reports 1996, 2000, Floyd et al, 2006). Targeted seeding of high-risk areas also was conducted after two other fires, but not until one or two years later ("delayed seeding" below): the fires in 2002 and 2003 were not seeded until 2004. This preliminary information has been widely used via accessibility through the NPS web site (http://www.nps.gov/archive/meve/research/weedrisk_assessment.htm and the forest health and fire planning "FRAME project" http://www.mpcer.nau.edu/frame/ (Turner, et al. 2006). The results of this project have been further used in the current revisions of Mesa Verde National Park's Fire Management Plan.

These aerial seeding treatments on MVNP and UMU lands appeared successful in that they dramatically reduced short-term influx of invasive plants (Floyd et al.

2006). Aerial seeding also has been shown to be effective in reducing erosion after fire in at least some settings (Groen and Woods 2008, but also see Kelley et al. 2005). However, concerns about mitigation using aerial seeding have been voiced, notably the possibility that the seeded plants may impede recovery of the local native flora (e.g. Keeley 2006, Robichaud et al. 2000) and that seed mixes may inadvertently introduce invasive species (such as occurred following the Cerro Grande Fire in New Mexico, Craig Allen, personal communication, Keeley et al. 2005). We also raised the question of how fuel structures and potential fire behavior may differ between seeded and unseeded burned areas. The long-term monitoring of BAER-sponsored fire mitigation treatments that is needed to evaluate these concerns has rarely been accomplished. In MVNP and UMU, the "immediate seeding" treatments had last been monitored in 2004, two years post-treatment, and no systematic monitoring of the "delayed seeding" had been conducted prior to the present study. The history of fires and mitigation treatments in MVNP and UMU, combined with our previous monitoring of seeded and unseeded areas, provided the opportunity for an experimental design comparing the effects of aerial seeding treatments (a) in similar habitats but in different years followed by different post-fire climatic conditions, i.e., drought or normal precipitation; and (b) in similar habitats that received "immediate seeding" after fire vs. others that received "delayed seeding" treatments.

Cheatgrass is an invasive weed of particular concern because of its well-known propensity to promote frequent and extensive burning (e.g., Billings 1990, Brooks et al. 2004, D'Antonio and Vitousek 1992, Link and Hill 2005, Link et al. 2006, Mack 1981, Whisenant 1990). This non-native annual grass has become increasingly prevalent in the Mesa Verde region during the past decade, especially in disturbed areas. For example, we observed relatively little cheatgrass after the 1989 fire in MVNP, but cheatgrass quickly dominated portions of the two fires from the year 2000, at elevations and substrates comparable to those of the 1989 fire. Using a spatially-explicit dynamic landscape model (SIMPPLE), the FRAME project determined that the centuries-long prehistoric fire rotation in MVNP could be reduced to ca. 45 years if cheatgrass expands as projected (Turner et al. 2009, MVNP Draft Fire Management Plan Appendix). Thus, characterization of habitats and disturbance processes that are susceptible to cheatgrass invasion within piñon-juniper and interior chaparral landscapes of Mesa Verde and the Four-Corners region has become increasingly urgent.

The extensive piñon mortality of 2002-2004 in the Four-Corners region raised a new concern about whether the open spaces beneath trees killed by drought or bark beetles will be invaded by cheatgrass. Selby (2005) observed that the frequency of cheatgrass was not correlated with percent piñon mortality after a widespread mortality event on the Uncompahgre Plateau in western Colorado. However, cheatgrass in her study area had been widespread and relatively abundant even before the regional piñon mortality event, and she sampled very soon after the piñons died, possibly before cheatgrass was able to spread into these newly disturbed sites. Therefore, we tested the idea that piñon mortality may in fact promote cheatgrass expansion in a different kind of ecosystem (Mesa Verde) where cheatgrass is still very uncommon in undisturbed areas, and we sampled several years after the mortality event in case cheatgrass may require some time to expand into this suitable new habitat.

Finally, there is considerable concern on the part of some resource managers that the introduction of aggressive, albeit native, perennial grasses may complete with native forbs. While the BAER objectives were to reduce non-native plants by the introduction of native grasses, it is conceivable that such introduction might reduce the possibility of other native species entering into the burned habitats. Opinions vary concerning this issue: natives may outcompete invasive species (e.g. Kennedy et al. 2002, Tilman 1999), but others have demonstrated that as diversity of natives increases so does diversity of invasive species (Stohlgren et al 2003). We address this potential problem by investigating patterns of floristic diversity and abundance of natives and invasive plants in seeded and non-seeded treatment areas in three recently burned areas in Mesa Verde National Park.

## KEY FINDINGS

### Question 1: *How have fuel profiles, potential fire behavior, and woodland structure in unburned piñon-juniper woodlands changed since the 2002-2004 mortality event?*

### Rationale:

Prior to the recent mortality event, crown fires occurred under extreme weather conditions in the naturally dense canopies and fuel conditions of persistent piñon-juniper woodlands across Mesa Verde. We recently documented a substantial decrease in density, basal area (Figure 1), and *live* tree canopy in piñon-juniper woodlands following a regional mortality event (Breshears et al. 2005, Floyd et al 2009). Such canopy opening may or may not alter future fire behavior, depending on the effect of *dead* fuels and the resultant growth of understory non-target layers (which may rapidly contribute to ladder fuels) on fuel continuity. Therefore, we investigated whether both horizontal and vertical continuity have changed significantly following the thinning from *Ips confusus* (piñon) and various insect or drought-related death (juniper), hereafter called "mortality event". Our current sampling started six years after the mortality waned, therefore needles and small branches were no longer present on trees. Most of the junipers remain rooted, but by 2010, dead piñons had begun to rot and fall to the ground.

How different is the woodland structure today, relative to the predominant structures of the 1990s? On Mesa Verde before the mortality event, seemingly continuous old-growth woodlands were broken up by natural discontinuities such as sandstone cliffs and punctuated by openings caused by numerous small and large disturbances. Smaller disturbances included patches of piñons killed by *Leptographium wagenerii*, black stain root disease, which has been active in Mesa Verde National Park since at least the 1930s (Kearns and Jacobi 2005), single-tree fires (Omi and Emrick 1980), and gaps created by deaths of individual trees from other causes. Using similar field sampling techniques to those used in the current study, between 1993 and 1995 we found that average tree cover in MVNP was 57.2%. By 2008, canopy cover in MVNP was reduced to approximately 40% primarily due to a significant reduction in cover of piñons killed by drought and beetles but also by drought effects on juniper (Breshears et al. 2005, Floyd et al. 2009). Canopy cover is useful in evaluating fuels and fire behavior therefore we repeated several canopy measurements in the current study. To further tease apart the fuel contributions in the post-mortality woodland, we investigated the structure (cover and height) of all woody species across piñon-juniper woodlands that were subjected to no, low and high mortality. We also analyzed the contribution of other factors such as slope and aspect on the fuel characteristics in the woodlands.

In the aftermath of a high-severity insect outbreak or drought-induced mortality event in a forest or woodland, the conspicuous dead trees can give an impression that fuel loads and fire potential have been dramatically increased. However, quantitative measurements of fuels and potential fire behavior lead to more nuanced interpretations. Hicke et al. (2012) recently reviewed the relevant literature on this topic, which comprised 39 published studies. From these studies and their own understanding of insect effects and fire behavior, they produced a conceptual framework for expected effects of relatively severe insect-caused tree mortality on fuels conditions and potential fire behavior (Table 1). Hicke et al. (2012) pointed out that the great majority of studies to date have focused on relatively mesic forests at higher elevations, e.g., lodgepole pine and spruce-fir forests. Piñon-juniper woodlands are conspicuously under-represented in the published literature, with only one previous study uncovered by their review (Clifford et al. 2008). Hicke et al. (2012) emphasized the need for studies in more xerophytic forest and woodland types because of important differences in fuels, canopy structure, and climate.

In order to further characterize the structural diversity of woodland stands before and after the mortality event we adopted a method used by Dordel et al on lodgepole pine forests. Dordel et al. (2008) computed a Shannon-Wiener Index for this purpose in their study of changes in lodgepole pine forest structure after mountain pine beetle outbreaks in the Canadian Rockies. Shannon-Wiener Indexes computed for height class, DBH class, DBH/height class, and species composition were greater in stands affected by outbreaks 15 years previously, but indices did not differ in stands in which the outbreak had occurred 25 or 65 years previously. We computed similar diversity indices in three representative woodland stands in MEVE that had been affected by the regional mortality event six years previously. The three stands differed in density and environmental context prior to the mortality event (Table 2), but all three represented common old-growth stand structures in the region, and all three experienced substantial mortality in 2002-2004.

Fuel load data were assembled into custom surface fuel models for input in fire behavior models. We used a well-established fire behavior model, NEXUS (Scott and Reinhardt 2001), which is based on Rothermel's surface fire spread (Rothermel 1972) and crown fire spread models (Rothermel 1991), as well as Van Wagner's crown fire initiation model (Van Wagner 1977). Use of such models to assess crown fire potential has been criticized on theoretical and practical grounds (e.g. Cruz and Alexander 2010), and we recognize these criticisms. Our surface fuel models are uncalibrated, and piñon-juniper woodlands present an especially difficult case for generating custom fuel models because of the lack of a clear separation of surface fuels and canopy fuels. We acknowledge that our fire behavior predictions may not perfectly represent actual fire behavior in the field, and we therefore emphasize in our discussion that: 1) the fuel load and forest structure estimates themselves, with the fire behavior predictions stemming from these estimates as secondary, to assist with interpretation, and 2) relative differences in fire behavior between sites, rather than absolute estimates of rate of spread, for example, or crowning and torching indices.

## **Methods**:

In 1993-1995 we determined fuel structures in 46 locations within the piñon-juniper woodland at MVNP using Brown's planar intercept method (Brown 1974, Brown et al. 1982) and measures of aboveground live biomass (Chojnacky 1994). In the current study, we overlaid these locations with the fire perimeters in a GIS environment to identify nine of original sampling sites that remained unburned. We returned to those sites in 2010 or 2011 and re-sampled the fuels using the same methods as were used previously and added new methods described below to more fully evaluate fuel structures.

Sample locations were selected using a stratified random approach. Using ArcGIS, we identified fire history polygons (either from historic UMU and MEVE fire outlines or from previous determinations by Floyd et al. 2000 and 2004), and generated random samples in each fire category. On the UMU Tribal land, fire dates were unknown except recent fires. Those unknown fire dates were treated as another stratum. The locations of 115 points (60 unburned and 55 from more recent burns) sampled during the 2010 and 2011 field seasons are shown, with the study area outline, in Figure 2). To address Question #1, the piñon-juniper woodland points that had not burned in the last 200 years were selected. These unburned woodlands represented types of piñon-juniper woodlands with varying understory characteristics and mature petran chaparral stands with scattered trees; only the 38 old growth piñon juniper stands were chosen for Question #1. Using a visual inspection, each site was characterized as to the degree of infestation by *Ips confusus* or direct effects of mortality: no mortality, low mortality (less than 10% tree mortality, few scattered dead trees), or the most common, high mortality (greater than 10% tree mortality, many dead trees).

We added to the traditional Brown's planar intercept (Brown et al. 1982) and conventional measures of above ground fuel biomass (e.g. Chojnacky 1994) a new sampling procedures to better evaluate vertical distribution of crown biomass, and canopy bulk density (CBD), which represents the mass of small diameter plant material that would burn in a crown fire. At each sample point a 50m transect was extended east and west. A 25m transect crossed perpendicular in the north and south directions from the center point. The east/west transects were divided into two 25m transects (Figure 3). Dead/down woody fuels were inventoried along these transects using a modified Brown's Planar Intercept method in which 1-hr (0-.25in diam.), 10-hr (.25-1 inch diameter), 100-hr (1-3 inch diameter), and 1000hr (3+ inch diameter) fuels were recorded along each transect, the transect being an imaginary plane from the ground to infinity. Any dead/down woody fuel (woody material detached on the ground or uprooted and at a $\geq$45 degree angle from vertical) that crossed this plane was recorded. Inventory of fuels began on east/west transects at the 5m mark to account for trampling around the center. 1-hr and 10-hr fuels were tallied from the 5m to 7m mark. 100-hr fuels were tallied from the 5m to 10m mark. 1000-hr fuels were recorded from the 5m mark to the 25m mark. The diameter of each 1000-hr fuel that crossed the plane was taken perpendicular from where the plane first intercepts the log. Identical methods were used for the next transect starting at the 30m mark and ending at the 50m mark (1-hr and 10-hr tallied from 30m to 32m, 100-hr tallied from 30m to 35m, and 1000-hr diameters were recorded from 30m to 50m). After the four transects running east/west were completed the pieces of fuel recorded were totaled. The target number for pieces inventoried was 100 pieces of fuel. If this number was not reached the methods were repeated on transects running north/south.

Canopy and surface fuel bed depth measurements were taken along the same east/west transects as the dead and down woody fuels. At 4-meter intervals (4m mark, 8m mark, 12m mark, etc.) along both the east and west transects, five pieces of information about the fuel beds was recorded: 1) litter depth (cm), 2) surface fuel depth (cm), 3) surface fuel type(LW=live woody, LH=live herbaceous, DW=dead woody), 4) canopy base height (m), 5) canopy height. We used 6 hemispherical photographic images per site to record leaf area index and canopy gap fraction (Keane et al. 2005) and analyzed the values with GAP Light Analyzer program to obtain "canopy openness" (Frazer et al. 1999).

Line intercept sampling (Mueller-Dombois and Ellenburg 2002) took place on the center transect, 50 m in either direction from the sample point, for a total of 100 m. The beginning and endpoints of each woody species was recorded.

Stand structural data were collected in six 100m$^2$ plots at each sample location (Figure 2). Within each plot we measured the diameter at the base (root crown) of each live tree, averaging two perpendicular measures for trees with irregular shape. An increment core sample was collected from each piñon. We did not attempt to age junipers due to inability to cross-date this species accurately.

Using a 0.25-m$^2$ point-frame sampling quadrat repeated six times at each sample location, we measured the relative cover of all shrubs, graminoids, and forbs at each sample point. ANOVA was performed using percent cover of each plant type (transformed with arcsin square root transformation to normalize data) as the dependent variables, and treatment (burned seeded, burned unseeded, unburned) as the independent variable.

We computed two diversity indices in each stand: *Simpson's Index of Dominance* and the *Shannon Index* (Odum 1971, page 144; see formulae in Table 2). These indices were originally developed for description of species diversity within a biotic community, not for structural diversity in a woodland. When applying the indices in this novel context, decisions about what constitutes a "species" or structural category can strongly influence the results of the computation. We followed Dordel et al. (2008) in treating each diameter class as a different "species" or structural component, analogous to a unique species in a community. However, we went beyond their analysis by classifying the structural components after the mortality event in two different ways. For the first method, we simply removed all of the dead trees, and computed diversity indices for the diameter classes represented in the remaining live trees. (We used basal diameters, and the classes were $\leq$1 cm, 1.1 - 5.0 cm, 5.1 – 10.0 cm, etc.). We also differentiated between tree species, i.e., we treated each diameter class of piñon trees as a different structural element than the same diameter class of juniper trees. For the second method, we treated the dead trees in each diameter class as new and unique structural elements, e.g., live juniper trees with diameters of 1.1 - 5.0 cm were one structural element, and dead juniper trees with diameters of 1.1 - 5.0 cm were a different structural element.

Another important difference between our approach and that of Dordel (2008) was in the selection of "control" stands. Rather than comparing stands that were subjected to the mortality event with stands elsewhere that were not affected, we reconstructed the pre-mortality structure of our stands by adding all of the dead trees, by species and diameter class, to the live trees that were still extant when the stands were sampled in 2008. (We excluded any trees that obviously had died prior to the mortality event – these were few in number and readily distinguished in the field from trees that had died in 2002-2004). Thus, we were able to compute structural diversity before and after the mortality event within the same stands.

The *Dominance Index* is actually a measure of the concentration of dominance within a few "species" (or classes of structural elements in our analysis). Thus, an increase in the Dominance Index is interpreted as a decrease in overall structural diversity. Consider, for example, a stand in which 91 of 100 trees are in a single diameter class, while nine other diameter classes are each represented by a single tree. Dominance in this stand is overwhelmingly concentrated in the one diameter class, and its Dominance Index is therefore greater than the Dominance Index of a stand in which the same ten diameter classes each contain 10 trees. The Dominance Index also is greater in a stand having fewer diameter classes than in a stand having more diameter classes, given the same distribution of trees among size classes in each stand, because dominance is more concentrated in the stand with fewer diameter classes.

The *Shannon Index* integrates both the number of "species" (or classes of structural elements in our analysis) and the distribution of individuals among those classes. It can be understood as a measure of the uncertainty as to which class would be selected in a random draw of individuals from the community. Uncertainty is greater if there are more classes to be drawn from and if individuals are more evenly distributed among those classes; uncertainty is lower if there are fewer classes with greater concentration of dominance within one or a few classes. Greater uncertainty in this context represents greater diversity within the community or the stand.

Finally, the fuels parameters measured at each site were used to predict potential fire behavior (surface fire rate of spread, surface fire intensity, torching index and crowning index) under extreme fire weather conditions. These predictions were compared among mortality categories and time-since-fire categories, emphasizing the relative differences between sites rather than the absolute estimates of fire behavior.

Custom fuel models were generated by modifying the TU1 model of Scott and Burgan (2005). The TU1 model represents conifer litter with herbaceous and shrub fuels in a dry climate, similar to Mesa Verde woodlands, so the surface area to volume ratios and the moisture of extinction (the dead fuel moisture at which the fire stops spreading) should adequately represent our study area. The other fuel model parameters in TUI were replaced on a site-by-site basis (one custom fuel model per site) as follows:

- Dead and down woody fuel loads (1-hr, 10-hr, 100-hr) were calculated from Brown's transects data. Litter load, based on litter depth measurements and the FIREMON-recommended bulk density value, was included with the 1-hr fuels.
- Live herbaceous and live woody fuel loads were estimated from cover and depth field measurements and the FIREMON-recommended bulk density values.
- Fuel bed depth was an average of point-intercept fuel bed depth measurements.

Forest structure and canopy fuel bed parameters were also estimated on an individual site basis from field data. Canopy fuel load was estimated by applying allometric equations to stand structure data (Miller et al. 1981). Canopy height and canopy base height were averages computed from point-intercept measurements. Canopy bulk density was calculated simply as the canopy fuel load distributed over the volume of space between the canopy height and the canopy base height. Forest cover, which was used to estimate wind reduction factor, was assessed based on the canopy cover from the line intercept measurements.

Custom surface and canopy fuel measurements were input into the NEXUS fire behavior model. The weather/fuel moisture scenario, meant to represent extreme fire weather as measured during recent fires such as the Bircher Fire, the Pony Fire, and the Chapin 5 fire, is provided in Table 3. Several iterations of model runs were performed for each site. First, to isolate the effects of changes to surface fuel loads on fire behavior, canopy cover was held constant (resulting in a standard wind reduction factor) and NEXUS was run in surface fire mode. Second, surface fire simulations were run while allowing the wind reduction factor to vary according to the actual canopy cover at a site. The second approach is reported in the results below. Finally, models were run in crown fire mode to allow torching and crowning indices to be calculated.

## **Results**:

Using ANOVA we tested if the cover of each species of shrubs or trees was significantly different across factors (level of mortality, soil classification, and aspect classes). There was a significant difference in piñon cover comparing degrees of mortality, with the lowest mean tree cover, 16%, in the areas with greatest mortality (Table 2, $F=3.6$, $p<.05$). When compared with areas of light mortality which averaged 43% cover, this is significant reduction in the canopy. At first glance, the 27 % cover of trees in sites that did not have appreciable mortality was surprising. However, these non-susceptible sites were open stands with relatively high density of perennial shrubs and fewer trees, such as on lower elevations on UMU Tribal land. These site conditions likely created differential susceptibility to beetles; drier more open sites, characterized by a dominance of perennial shrubs and a low cover of piñon and juniper, were less likely to have had the canopy characteristics in 2002 that would support a beetle infestation. Therefore it is more reasonable to statistically pair comparable low and high mortality sites to one another in order to see the fuel changes in susceptible sites. The causes of mortality on juniper are less clear; no particular insect has been identified as a single cause although *Trachykele blondeli*, the western cedar borer, has been recorded as a mortality agent in MVNP. Also, despite an 11% overall reduction in juniper density across the landscape during the mortality event (Floyd et al. 2009), we did not detect a reduction in average juniper cover among the categories of stand-level mortality (Table 4).

A second measure of cover using hemispherical photographs (Table 5) confirmed a significant difference in canopy closure (defined as 100% minus % canopy openness) across the mortality categories. In a subset of comparable locations on Chapin and Wetherill mesas in Mesa Verde National Park, sites that experienced intensive mortality were now more open, with 48.5% canopy cover, compared with those that experienced low levels of mortality, 58% canopy cover. While the trends in these values are similar to the line intercept sampling above (i.e. lower cover in high mortality sites), hemispherical photography includes all shrubs and trees so the magnitude of the canopy measures is greater.

Shannon-Weiner Indices are summarized in Table 2. In all three stands, the Dominance Index *increased* after the mortality event when all of the dead trees were removed from the analysis—meaning a concentration of dominance within fewer structural classes and a consequent *reduction* in overall structural diversity. This result and interpretation reflect the smaller number of classes of structural elements present after the mortality event and the corresponding greater concentration of dominance within each of the remaining structural classes. Most of the larger diameter classes had been represented by small numbers of trees before the mortality event such that death of even a few of these large trees meant that some classes were no longer present in the stand as live trees. However, the Dominance Index *decreased* after the mortality event when dead trees were treated as new structural elements—meaning an *increase* in structural diversity. This occurred because there were now more classes of structural elements than before, with correspondingly lower proportions of trees in each class; the concentration of dominance was therefore reduced and overall structural diversity increased.

The Shannon Index *decreased* in all three stands after the mortality event when all of the dead trees were removed—meaning a *reduction* in structural diversity. This was because fewer classes were present after the mortality event; consequently the uncertainty that would be associated with a random draw was reduced. However, the Shannon Index *increased* after the mortality event when dead trees were treated as new structural elements—meaning an *increase* in structural diversity. Treating dead trees of each species and diameter class as new and unique structural elements meant that each class now would have a lower probability of being drawn in a random selection of trees, thus, there would be more uncertainty as to which class would be drawn, meaning greater stand structural diversity.

### *Fuel structures*

Given the extent of mortality in piñon-juniper woodlands in southwest Colorado, it was difficult to find patches within the woodland that had no *Ips* activity or other causes of tree death unless the stands were very open and had few trees (such as at the lower elevations on UMU Tribal land). We located six largely unaffected stands in old-growth woodlands and compared the fuel structures to those with low and high mortality (Table 6). Fuels data did not meet the assumption of normality therefore we compared groups with the non-parametric Kruskal-Wallis test. There was a nearly significant difference in combined fuels from 1 to 100 hr classes when compared across the *Ips* categories, with the largest ranks in sites with greatest mortality ($X^2$=5.7, p=.057). Also, there was significantly greater 1000 hr fuels (rotten and sound combined $X^2$=8.2, p<.05) in high mortality stands.

Changes in ground cover after the mortality event were analyzed in 330 point frames and ANOVA was run to test for differences in invasive grasses across the mortality categories. Invasive grasses (cheatgrass and smooth brome) did not increase in low and high mortality centers when compared to intact woodland (F=1.18, p>.05). Similarly, invasive forbs were not favored in the sites where trees died (F=2.0, p>.05). This contrasts sharply with post-fire disturbance which favors invasives (see below) and may be due to little disturbance of the soils under dead trees.

### *Fire behavior*

Comparing NEXUS surface fire behavior predictions using the 2010-2011 samples (surface fire simulations for runs which allow the wind reduction factor to vary according to the actual canopy cover at a site) separated into stands with no mortality, low mortality and high mortality, we detected no differences in spread rate, fireline intensity, flame length, heat per unit area, or torching index when we compared stands across the Mesa Verde landscape in 2010-11 (Table 7). We also saw a difference in crowning index (the wind speed needed to carry the fire through the canopy), with higher values in the no-mortality sites. However, these stands generally had a very open structure even before the mortality event and were concentrated on drier, less productive lower-elevation sites. Such stands had low tree density and high shrub components and were not targets for beetle infestation, hence they cannot be used as a reliable baseline for changes in fire behavior that might be attributed to the mortality event in denser stands on more productive sites.

Therefore, to better assess fire behavior before the mortality event in Mesa Verde's piñon-juniper woodlands, we used fuel data collected in 1993 at six of the nine unburned sample locations on Chapin and Wetherill mesas in Mesa Verde National Park, comparing values with repeated sampling in 2010-2011. (Of the nine locations re-sampled in 2010-11, three had missing 1993 data and were discarded for this analysis). It should be noted that the input for NEXUS models run on the 1993 data required adjustments to the "fuel depth" measurements to make the 1993 and 2010-11 samples comparable, and the sample size of appropriate 1993 sites is low. Only surface fire models could be run due to a lack of certainty surrounding canopy base height and crown bulk density in the 1993 measurements. Thus, 1993 represents the fire behavior before the mortality event (pre-condition) and 2010-11 represents the fire behavior after the mortality event. While no differences were detected in spread rate or flame lengths, there was a significant difference in heat per unit area, with far greater average heat produced after the mortality event (Table 8).

## **Discussion:**

We used a somewhat novel approach to characterize the changes in stand structural diversity following a major mortality event in old-growth woodland stands of MVNP. By comparing current woodland structure among stands of varying degrees of infestation, we detected a significant thinning of the woodland canopy by two different metrics (gap photographs and line intercept sampling). When compared with stand canopy characteristics in 1993, before the mortality event, today's canopy is reduced by approximately one-third of the original canopy cover. Piñon mortality due to *Ips* beetle infestation from 2002-2004 accounts for most of these changes, however juniper and possibly piñon mortality from drought-related causes contribute as well. Another measure of woodland structural change came from the Shannon Wiener calculations of pre and post mortality stand reconstructions. Both the Dominance Index and the Shannon Index indicated a reduction in structural diversity if the dead trees were simply removed from the analysis, but both indices indicated an increase in structural diversity if the dead trees were regarded as new structural elements in the stands. Our analysis in which we simply removed the dead trees is probably the more relevant method from a long-term perspective, because nearly all of the dead piñon trees will likely fall within the next decade (Kearns and Jacobi 2005). Thus we conclude that structural diversity in old-growth piñon-juniper woodlands is increased temporarily (a few years at most) by the mortality event, but is reduced over the long term. This effect of insect-caused mortality on stand structural diversity in piñon-juniper woodlands differs from the effect found by Dordel et al. (2008) in lodgepole pine forests, where structural diversity was greater in stands 15 years after mountain pine beetle outbreak. The difference probably reflects the different stand structures prior to the insect outbreak. Lodgepole pine forests tend to be dominated by a single canopy stratum, which is "opened up" by beetle-caused mortality, allowing suppressed subcanopy and understory trees to grow faster and assume greater dominance. Whereas piñon-juniper woodlands have a diverse structure with many tree sizes well represented before the mortality event, and the insects selectively remove many or most of the larger size classes.

In summary, all measures (density, cover, and structural diversity) suggest that following the beetle infestation, the piñon-juniper woodland on Mesa Verde has become *considerably less dense with lower cover and declining structural complexity*. Next we turn our attention to the possible effect that such structural change might have on future fire behavior.

Fire behavior models, while imperfect quantitative predictors of fire behavior (Cruz and Alexander 2010), nonetheless are useful in comparing stands across a landscape. Here we compare outputs that are likely attributed to the structural changes that occurred during the mortality event in the Mesa Verde region. We did this both by a *spatial* comparison of piñon juniper woodlands subjected to different levels of infestation, and a *tempora*l comparison of pre-mortality and post-mortality stands in nearby areas in Mesa Verde National Park. The temporal comparison suggested that while certain measures of surface fire behavior (spread rate, flame length) do not change across levels of infestation, heat per unit area is greater in high-mortality stands. This is consistent with greater levels of slower burning fuels (100-hr, litter). It is likely that with greater quantities of fuels in some categories, notably litter and 100-hr fuels, after the outbreak, there is potential for longer fire residence time and greater total heat release than before the outbreak. While 1000 hr fuels are not considered important influences of a flaming front and therefore are not entered into fire behavior models, nonetheless the significant increase in these larger fuels and greater residence time for fire may have ecological consequences, especially soil impacts. With the longer residence time to dry out canopy fuels, it is possible that post-outbreak stands might be more prone to torching fire behavior, even though a sparser canopy would make active crown fire less likely.

## **Question 2**: ***How do fuel profiles, potential fire behavior, plant diversity and structure differ between recently burned areas and across the fire chronosequence?***

### **Rationale:**

The fire chronosequence at Mesa Verde National Park presents a wide variety of fuels that vary with time-since-fire as well as with pre-fire vegetation. We predicted that pre-fire vegetation, particularly the presence of perennial sprouting species, will influence the structure of post-fire vegetation communities and fuel accumulation. In areas with vigorous sprouting perennial shrubs, biomass will accumulate quickly after fires and, invasive weeds will not be as successful as areas lacking these competitors. However, after stand-replacing fires in old-growth piñon-juniper woodlands, where perennial sprouters are typically lacking, field observations suggest that biomass accumulates slowly and invasive weeds are often prolific. These patterns will be reflected in fuel loads over the time-since-fire.

In this section we address how fuel structures change across the post-fire landscape following large stand-replacing wildfires that occurred on Mesa Verde recently and in the last two centuries, comparing fuel conditions with old-growth woodlands that have not burned for at least four centuries (Floyd et al., 2000, 2004).

### **Methods:**

The Mesa Verde landscape was divided into four categories of post-fire succession—early succession in recent fires (less than 25 years since fire), mid succession (25-200 years since fire), late succession (200-400 years), and old-growth

(>400 years)—to determine fuel characteristics across the landscape. As described in Question 1 above, we sampled 115 points across the fire chronosequence using the methods described in Question #1 above. We also investigated plant associations at each sample location. Two 10m by 10 m releve plots were established at each end of the transects and the cover/abundance of each plant species was recorded using the Braun-Blanquet scale (Mueller-Dombois and Ellenburg 2002). For this floristic analysis we lumped the two oldest categories used in the fuel sampling (late succession and old-growth). Cluster Analysis and TWINSPAN of early, mid and late successional groups were used to group plots by their floristic similarities. Each successional group was subjected to ordination using Nonlinear MultiDimensional Scaling (NMS, McCune and Grace 2002). Here we will discuss floristic trends in the post-fire sequence as they relate to fuel accumulation.

## **Results:**

We identified 178 plant species in the 230 study plots and grouped them into twenty plant communities. Nine associations were defined in the early succession category, six in mid succession and five in late succession category. These will be summarized below as they pertain to fuel structures.

Late succession stands (>200 years since fire) were dominated by *Pinus edulis* and *Juniperus osteosperma* with differing degrees of perennial shrubs, herbaceous forbs and perennial bunchgrasses in the understory. Some of these stands at the lower elevations on UMU included standing and down burned juniper snags, but others had no visible evidence of past fire (Floyd et al. 2000).

Mid-succession stands (20-200 years since fire) were dominated by numerous tall shrubs, with varying grasses in the understory. In the northern portions of the study area (Figure 3) *Amelanchier utahensis*, serviceberry, *Purshia tridentate,* bitterbrush and *Quercus gambelii,* gambels oak dominated, while in the more southern, lower elevations, *Artemisia tridentata* dominated.

We are especially interested in the presence and abundance of actinorhizal species, woody shrubs which have symbiotic associations with the nitrogen fixing *Frankia sp.* (Busse and Riegel 2009, Neary et al. 1999, Oakley et al. 2003, Paschke 1997). These species may contribute to soil nitrogen dynamics after fire and may have a disproportionate influence on biomass accumulation and fuel accumulation. We did not detect a statistically significant trend in the relative contribution of actinorhizal perennial shrub species across the fire chronosequence. Actinorhizal *Purshia tridentata.* bitterbrush, and *Cercocarpus montanus*, mountain mahogany, were found throughout the five vegetation associations in the mid-succession category where they can be the dominant species (e.g. the 1934 fire) and tend to have a lesser presence in older successional stands and early post-fire stages (Figure 4).

The diverse early succession category included recent fires from the past 25 years; portions of the recent fires in 1996, 2000, 2003, and 2005 support dense perennial shrub stands that sprouted soon after the fires while others support grass-forb communities. Grass-forb stages had varying amounts of native grasses, native forbs, invasive forbs such as muskthistle, and invasive grasses such as cheatgrass. Most of the native grasses in the early succession stages were perennial bunchgrasses; *Pascopyron smithii* is the most spreading, rhizomatous species among the native grasses, and dominated one of the communities. The other native perennial bunchgrasses, such as *Elymus elymoides*, *Oryzopsis hymenoides, Agropyron trachycaulon, Stipa comata* and others do not create a continuous flammable fuel. Rather, spaces of bare soil or forbs occur between individuals. Thus the recent fires support considerable variation in fuel continuity.

### *Patterns of invasive species in the fire chronosequence*

There was a significant difference in the mean cover of invasive grasses, primarily *Bromus tectorum*, cheatgrass, and *Bromus inermis*, smooth brome, across the fire chronosequence, F=8.62, p<.01 (Figure 5). A similar trend was seen for invasive forbs, primarily *Carduus nutans*, muskthistle, with *Salsola iberica* and other forbs in smaller quantities and cover, F=9.79, p<.01 (Figure 5). Muskthistle invaded the 1996, 2000 and 2002 fires with extremely high densities and cover in areas where competitive natives were lacking (Floyd et al. 2006); we are encouraged to see a decline cover of this species over time (although muskthistle establishment fluctuates with precipitation patterns therefore caution should be taken in interpreting trends). On the other hand, cheatgrass cover is increasing. Below, in Question 3, we examine further the responses of cheatgrass and muskthistle to post-fire mitigation in recent fires. Invasive plant species were conspicuously absent in mature piñon-juniper woodlands yet commonly dominated recent burns.

### *Effect of time since fire on fuels and fire behavior*

Fuels data across the fire chronosequence were used in NEXUS fire models to predict potential patterns in fire behavior. The stands were grouped into early, mid, and late successional categories by fire dates, and compared with old growth stands. Significant differences in each measure of surface fire behavior (spread rate, fireline intensity, and heat per unit area) were detected. Recent fires (2000 and more recent) have not developed significant fuel accumulations and fire behavior values are relatively. The spike in fire behavior in mid-successional stands can be attributed to fuel accumulations in woodlands which burned between1800 and the early 1990's (Table 9). As previously discussed, vigorous resprouting of perennial shrub layers contribute to fuel continuity and depth. The fire behavior changes in the later successional stages (>200 years since fire) with a reduction in all measures of fire behavior, and is especially low in the oldest piñon juniper woodlands—those that have not burned for over 400 years. Crown fire behavior was not analyzed along the post-fire chronosequence because 3 of the 4 categories (all but the old-growth stands) lacked sufficient canopy trees.

## **Discussion**:

On the Mesa Verde cuesta, the response of plant communities after fire varied significantly, in a large part based on whether the pre-fire vegetation consisted of a significant shrub component. Fuel and floristic data collected in recent fires vary widely across Mesa Verde depending on pre-fire vegetation and susceptibility to invasive plant species, especially cheatgrass. Where cheatgrass dominates, as it does in several of the recent fires, it would promote fast-moving, low intensity fires. Such fires would be out of the HRV in this region. However, if extensive fires do not occur and woody species are able to establish (as has happened in the past centuries), mid-successional stands develop with rich perennial shrublands. This has implications for post-fire fuel accumulation and explains in part the spike in predicted fire behavior in the mid-successional sere (Table 9). With additional fire-free periods, mature woodlands develop, characterized by a dense canopy and a decline in perennial shrubs; measures of fire behavior are reduced in these woodlands. This may seem counterintuitive at first glance, however further consideration suggests that as the stands age, the tree layer and shading increases, biomass of the understory, especially in the perennial shrub layer declines. Old-growth piñon juniper stands develop a rich herbaceous understory of native forbs and grasses and a diverse biotic crust community. *Purshia tridentata* is the dominant shrub in northern mesas in MVNP while *Artemisia tridentata* dominates on the southern portions of the study area on UMU tribal lands and in canyon bottoms but shrub biomass is far less than in mid succession woodlands. Hence contribution to fuel depth from the understory declines as piñon and juniper crowns increase and canopy closes. Further maturation of the old-growth woodland is punctuated by gaps caused by the deaths of small patches of trees (by small or single tree fires, by native pathogens such as beetles and black stain root disease, by other disturbances, and by senescence) and the canopy closure is reduced but surface fuels remain minimal. Our findings regarding fuels conditions and potential fire behavior in old stands help to explain the occurrence of ancient, persistent piñon-juniper woodlands on the Colorado Plateau and elsewhere (Romme et al. 2009).

## **Question 3**: ***How does community composition differ between recently burned areas that were or were not treated with post-fire seed applications, with an emphasis on comparing abundance and richness of native vs. non-native species?***

### **Rationale**:

Our previous work suggests that post-fire aerial seeding treatments (Table 10) on MVNP and UMU lands were successful in that they dramatically reduced short-term influx of invasive forbs (Floyd et al. 2006). Long-term trends are not known. However, concerns about mitigation using aerial seeding have been voiced, notably the possibility that the seeded plants may impede recovery of the local native flora and that additional weeds may be introduced via contaminated seed mixtures (e.g. Craig Allen personal communication, Hunter et al. 2006, Keeley et al. 2003,2005, Keeley 2006, Robichaud et al. 2000). In this study we addressed these concerns by comparing abundance and species richness of native and non-native species in areas that were seeded vs. unseeded. We also compared areas that were seeded immediately after the 1996 and 2000 fires vs. delayed seeding following the 2002 and 2003 fires.

### **Methods:**

Two different approaches were taken to sampling recent fires in MVNP and UMU Tribal land to test the effectiveness of post-fire seeding treatments. First, we sampled a series of plots in areas burned in 1996, 2000, 2002, 2003, 2005, and 2008, comparing the diversity and abundance of native and invasive species in a random sampling of post-fire seeding treatments with a random sample of unseeded control areas. Using independent samples T-tests and One-Way ANOVA, we compared the mean values of these four variables across treatments. We also compared diversity and abundance of native and invasives in "immediate treatments" in the 2000 and 2002 fires in Mesa Verde National Park vs. "delayed treatments" following the 2002 and 2003 fires.

In our second approach, we paired a series of plots, matching seeded and unseeded plots within close proximity and therefore nearly identical soil properties, precipitation patterns, microclimate and potential seed sources. This was done only in the 1996, 2000 or 2002 fires that occurred in MVNP. Either three or four 9 $m^2$ plots were established per sample area for a total of 26 sample pairs. Again, we measured native and invasive species diversity and abundance in each plot.

## **Results**:

The mean cover of native grasses was approximately twice as high in randomly placed seeded plots compared with untreated plots (Table 10), demonstrating that the seeded native perennial grasses were well established. No significant difference in the cover of native forbs was detected among treatments, suggesting that the spaces between the native perennial grasses has allowed infill of native forb species and has not presented an aggressive competitive situation.

The mean cover of invasive grasses was reduced by three-fold (Table 11). While our previous studies have shown up to an 18-fold reduction of *Carduus nutans*, musk thistle, *Cirsium arvense*, Canada thistle, and other invasive forbs, a decline we previously attributed to post-fire seeding treatments (Floyd et al. 2006), we did not detect any differences in these species across treatments in 2010-2011.

Were these effects the same for "immediate" and "delayed" seeding? In MVNP, the 1996 Chapin 5 Fire was seeded within a month and native seed application was followed immediately by snowfall and a winter with excellent snow. The two fires that occurred in 2000 were also seeded immediately after the burns. However, the 2002 and 2003 fires were seeded in 2004 (BAER Reports 1996, 2000 and NPS Reports for 2002 and 2003). "Immediate" treatments had significantly lower abundance of invasive grasses (Table 12, $F=16.14$, $p<.01$) and invasive forbs ($F=6.39$, $p<.05$) when compared with "delayed" treatments.

Paired plots showed trends similar to randomly placed plots discussed above. In the 26 paired 9 $m^2$ plots, total plant diversity was nearly identical across the treatments (Table 13). However, a significant difference was found in the abundance of invasive plants across the treatments. Invasive plant abundance was much lower in seeded plots than untreated paired plots ($T=2.6$, $p< 0.05$) (Table 14). Specifically, cheatgrass was three-fold lower in seeded areas, with mean abundance in seeded areas of 32 plants/9$m^2$ plot and 100 plants/9$m^2$ plot in unseeded areas ($T= 2.32$, $p=<.05$). No statistical difference was found in the abundance of musk thistle, between seeded and unseeded treatments. Using NEXUS fire behavior models we did not detect differences in predicted fire behavior comparing the paired plots. There was no difference detected in mean spread rate ($F=2.14$, $p>.05$), heat per unit area ($F=2.42$, $p>.05$), fireline intensity ($F=3.56$, $p>.05$) or flame length ($F=3.63$, $p>.05$). However, this result is likely misleading because NEXUS assumes a homogeneous fuel bed and does not explicitly incorporate the effects of fuel continuity. The most important difference between a fuelbed of dry cheatgrass vs. native herbs is not related to fuel mass but to the typically greater continuity of the cheatgrass fuel. A fuelbed of dry cheatgrass probably resembles the idealized fuelbed assumed in the model, whereas a native fuelbed usually is far more heterogeneous and patchy. Thus, our fire behavior modeling results probably are relatively realistic for cheatgrass but far less realistic for a native herbaceous fuel bed. Actual fire spread typically is less in a native fuelbed than in a fuelbed dominated by cheatgrass (Link 2006).

## **Discussion**:

Our long-term analysis of seeding effectiveness suggests that introduction of native perennial grass seeds at 5-8 lbs per acre (BAER Reports 196, 200a, 2000b) effectively reduces, but does not eliminate invasions of non-native plant species from the post-fire environment. Judging from the 1996 treatments, this reduction persists for at least 15 years. While there are several species of invasive, non-native plants present in the post-fire successional environment on Mesa Verde, we suggest that the most problematic invasive is cheatgrass. This aggressive grass germinates in the fall months especially during wet years, flowers early in spring and creates a dry, continuous and highly flammable mat of fuel by early summer that is unprecedented in this environment. Introduction of native perennial grasses reduces the cover of invasive grasses and forbs and is most pronounced in treatments where the seeding treatment takes place in the same year as the fire. It is encouraging that musk thistle appears to be declining from the post-fire landscape. Musk thistle was the most prolific weed after the 1996 and 2000 fires (Floyd et al. 2006), but its effect on the landscape, particularly future fire behavior appears to have lessened. Mesa Verde National Park and several federal and state agencies have introduced biological controls such as the seed head weevil in the region. While we may be seeing the effect of those controls in Mesa Verde, along with the natural “in-fill” of grasses stimulated by the aerial seeding, trends must be treated with caution because musk thistle germination is heavily tied to precipitation trends. Cheatgrass has gained dominance and its effect on future fire behavior is likely more critical.

While our data suggests that seeding immediately, rather than delaying treatment for a year or more is the most favorable means of reducing invasive species on Mesa Verde, these trends must be considered cautiously, particularly in light of differences in weather across years (Table 15).

## MANAGEMENT IMPLICATIONS

### (1) Fuels and fire hazard

The 2002-2004 mortality event has altered fuel structures in Mesa Verde's old-growth woodlands, but—contrary to what is sometimes assumed--these changes do not translate into a simple or linear increase in fire risk or fire hazard. Given typical extreme fire weather conditions, potential surface fire characteristics—rate of spread, fireline intensity, and flame length—have not changed significantly. Torching potential has not changed significantly. Small (1 to 100-hr) and large dead woody fuels (1000-hr fuels) have increased, which may lead to increased fire residence time, increased total heat release, and increased safety concerns for firefighters. However, the likelihood of active crown fire has probably been reduced in most stands. We were unable to test for this explicitly without pre- and post-*Ips* canopy data due to the confounding relationship between stand density and *Ips* severity (i.e. the no mortality sites were more open and less likely to carry crown fires, but also not likely to be attacked by beetles). Effectively, the Ips outbreak reduced the tree density in the densest stands such that we saw similar crowning indices in areas with low and high levels of mortality.

*These changes mean that large, wind-driven, active crown fires, the kinds of fires that historically have burned the greatest areas and been most difficult to control, are now somewhat less likely to occur in the old-growth woodlands of the Mesa Verde cuesta than they were before the mortality event. Future fires are more likely to be slower-moving surface fires, punctuated by individual tree crown torching; these may be somewhat more difficult to control under adverse weather conditions than were similar surface fires prior to the mortality event, and hazards for firefighters may be greater because of increased surface fuel loads.*

### (2) Old-growth stand structure

Tree density, basal area, canopy cover, and structural diversity have been generally reduced in old-growth piñon-juniper woodlands of the Mesa Verde cuesta. The greatest reductions have occurred in stands where density, basal area, and canopy cover were greatest before the mortality event; these typically were stands on the most productive sites. However, no novel stand structures have resulted from the mortality, i.e., lower-density stands like those produced by the mortality event were common on the cuesta even before the mortality event, primarily on less productive sites.

*Despite some substantial structural changes in some stands, all of the post-mortality stands that we examined appear to remain within the historical range of structural variation for old-growth piñon-juniper woodlands on the Mesa Verde cuesta. We recommend that these stands be considered for preservation whenever possible. Conserving the structural characteristics is critical to their maintenance, and conservation of these systems will maintain their rich native plant and animal biodiversity.*

### (3) Susceptibility to invasion by non-native plant species

Recently burned woodlands are highly susceptible to invasion by cheatgrass, musk thistle, and other non-native plant species. However, old-growth woodlands that have not burned for centuries appear relatively resistant to non-native invasion, so long as the soils are not scarified or otherwise disturbed.

*Active efforts to control non-native invasion will be beneficial in all or nearly all recently burned woodland areas. Avoiding or preventing soil disturbances in unburned old-growth woodlands will help to maintain the inherent resistance of these plant communities to non-native invasion.*

(4) <u>Efficacy of post-fire seeding of native grasses to reduce non-native invasion</u>

The density and cover of native grasses continues to be higher, and the density and cover of non-native grasses and forbs continues to be lower, in burned areas that were seeded with native perennial grasses immediately after wildfires that occurred as much as 15 years ago. Moreover, we observed no reduction in the diversity or abundance of native forbs in places that were seeded with native perennial grasses compared with areas that were not seeded.

*Immediate post-fire aerial seeding with native perennial grasses is an effective method for mitigating non-native invasion of burned areas, with no apparent adverse effects on natural successional processes, and beneficial results persisting for at least 15 years (and probably longer).*

## RELATIONSHIP TO OTHER RECENT FINDINGS AND ONGOING WORK

Many on-going studies are similarly investigating possible outcomes of drought and beetle infestions on future fire behavior, but as Hicke et al. (2012) points out, few have been in the piñon-juniper woodlands. Hicke et al. (2012) recently reviewed the relevant literature on this topic, which comprised 39 published studies. From these studies and their own understanding of insect effects and fire behavior, they produced a conceptual framework for expected effects of relatively severe insect-caused tree mortality on fuels conditions and potential fire behavior (Table 1). Hicke et al. (2012) pointed out that the great majority of studies to date have focused on relatively mesic forests at higher elevations, e.g., lodgepole pine and spruce-fir. Piñon-juniper woodlands are conspicuously under-represented in the published literature, with only one previous study uncovered by their review (Clifford et al. 2008). Hicke et al. (2012) emphasized the need for studies in more xerophytic forest and woodland types because of important differences in fuels, canopy structure, and climate. Hence, this study will augment current work in other forest types also affected by drought and beetle infestations.

Our studies of structural changes in piñon juniper woodlands following the 2002-2004 mortality event concluded that structural diversity in old-growth piñon-juniper woodlands is increased temporarily (a few years at most) by the mortality event, but is reduced over the long term. This effect of insect-caused mortality on stand structural diversity in piñon-juniper woodlands differs from the effect found by Dordel et al. (2008) in lodgepole pine forests, where structural diversity was greater in stands 15 years after mountain pine beetle outbreak. The difference probably reflects the different stand structures prior to the insect outbreak: lodgepole pine forests tend to be dominated by a single canopy stratum, which is “opened up” by beetle-caused mortality, allowing suppressed subcanopy and understory trees to grow faster and assume greater dominance, whereas piñon-juniper woodlands have a diverse structure with many tree sizes well represented before the mortality event, and the insects selectively remove many or most of the larger size classes.

## FUTURE WORK NEEDED

Many of the suggestive trends mentioned require or would be enhanced by further research. The decline in musk thistle from post-fire environments must be verified in future wet and dry years. Continued monitoring of invasive plant decline in seeded treatments is needed to verify whether the promising 15 year trends reported here are longer-lasting. As future fires occur, special attention should be paid to fire behavior and post-fire succession in areas currently dominated by cheatgrass.

Our fuel data might yield better predictive results with next-generation physics-based fire behavior models such as FIRETEC (Linn et al 2002) and Wildland Urban Interface Fire Dynamics Simulator (WFDS; Mell et al. 2007). These are able to overcome many of the limitations of traditional fire behavior models, including the assumptions of spatial homogeneity in fuel beds and temporal homogeneity in weather inputs. These models require detailed input data and may provide a potential for improved modeling in pinon-juinper systems, which violate many of the assumptions in the traditional models.

The principal investigators Floyd-Hanna and Crews are involved in a related study at Mesa Verde National Park and Arches and Canyonlands National Parks in which the possible influence of nitrogen deposition on post-fire successional trends is being investigated. These data should complement the current study, as atmospheric inputs of nitrogen may add to those from combustion of fuels and continually “fertilize” these ecosystems which developed under nitrogen limitation. This hypothesized effect on fire behavior will be investigated.

**DELIVERABLES CROSSWALK**

| **Deliverable** | **Description** | **Delivery Dates** | **Status** |
|---|---|---|---|
| Field tour | On site tour to inform NPS, BLM, and UMU managers of fuel inventory methods | September 2009 | ***Completed-Met with representatives of UMU, BIA and NPS at Mesa Verde National Park*** |
| Website | Update MVNP Website with year 1 results | October 2010 | ***Updated periodically*** |
| Report to JFSP | Year one annual report | September 30, 2010 | ***Completed September 2010*** |
| Workshop | Public and management oriented site visits on Chapin Mesa, MVNP | August 2011 | ***Completed- August, 2011 with over thirty participants*** |
| Report to JFSP | Year two annual report | September 30, 2011 | ***Completed*** |
| Published journal articles | Submit two articles to peer-reviewed journals such as Forest Ecology and Management, Ecological Applications and others | December 2013 | ***Manuscripts in preparation*** |
| Professional presentation | Presented this project at the Fire Ecology meetings in Santa Fe, NM | February 2012 | ***Completed*** |
| Visitor center or museum display | Create a display for the Mesa Verde National Park Museum that can be updated and linked with field tours | August 2012 (and ongoing) | ***Discussions with museum staff have been initiated*** |
| Website | Update links to weed risk model on MVNP website USGS southwest weed clearinghouse websites Link to SW Fire Science consortium | December 2011 (and ongoing) | ***Link is being developed to Southwest Fire Science consortium (A. Thode); contact made with other web site managers*** |
| Workshop/closeout | Present results to UMU and MEVE resource and fire professionals | June-August 2012 | ***Planned but exact dates pending*** |

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List of data files to be archived at USFS metadata site (sent to Laurie Porth)

## Canopy and Down Wood Characteristics

1993-2010-2011 Down woody fuels summary.xls

Down Woody Fuels with independent variables.xls

MVNP Structural Diversity Actual.xls

Shrub and tree cover line intercept sampling.xls

Calculations of Shannon W indices of structural diversity.xls

Ground fuels point frame.xls

## Fire Behavior Modeling Files

Surface Feb 22 2012.scn

Crown BCI.Feb 22 2012.scn

Surface.ctr.93 data. April 26 2012.scn

SEL_Custom fuel models_Feb 22 2012.fmd

Custom fuel models. 93 ata. Feb 22 2012.fmd

# Tables and Figures for Mesa Verde Report

Table 1. Conceptual framework for expected changes in fuels and fire behavior in forests after a severe or moderately severe insect-caused mortality event (modified from Fig. 2 in Hicke et al. 2012). The table focuses on the “gray stage” (after dead needles have fallen, 5-10 yr after the mortality event) which was the stage of the old-growth piñon-juniper stands in Mesa Verde National Park that were sampled in this study.

| **Fuel or fire behavior parameter** | **Condition in gray stage relative to pre-mortality condition** | **Level of confidence (based on number of studies and consistency of findings)** |
|---|---|---|
| Canopy bulk density | Large decrease | Relatively high |
| Fine surface fuel load | Moderate increase initially, then moderate decrease | Relatively low for initial increase, but relatively high for later decrease |
| Coarse surface fuels | Moderate increase | Relatively high |
| Ladder fuels (shrubs & small trees) | No change or small increase | Relatively low |
| Surface fire behavior – rate of spread, reaction intensity, & flame length | Moderate increase | Relatively high |
| Torching potential | No change or small increase | Relatively low |
| Active crown fire potential | Large decrease | Relatively high |

Table 2. Structural diversity indices in three piñon-juniper woodland stands in MVNP before and after a regional mortality event that occurred in 2002-2004. Post-event sampling was conducted in 2008. See text for definitions of structural elements. Indices were computed as follows (Odum 1971;144):

> **Dominance Index:** c = sum(Pi$^2$) … where Pi is the proportion of trees in each class of structural elements (i.e., in each unique combination of species, diameter, and live/dead status)
>
> **Shannon Index: H** = - sum(Pi*log(Pi))

| | **Before Mortality Event** | **After Mortality Event, *Dead Trees Deleted*** | **After Mortality Event, *Dead Trees Treated as Unique Structural Components*** |
|---|---|---|---|
| *Stand 1– near upper elevational limit of woodland, relatively low tree density prior to mortality event, 10% of trees died in the event* | | | |
| Number of Structural Components | 26 | 24 | 37 |
| Number of Trees | 200 | 179 | 200 |
| Dominance Index | 0.15 | 0.18 | 0.14 |
| Shannon Index | 1.06 | 1.00 | 1.16 |
| | | | |
| *Stand 2 – near middle elevational limit of woodland, relatively high density prior to mortality event, 33% of trees died in the event* | | | |
| Number of Structural Components | 23 | 22 | 37 |
| Number of Trees | 280 | 186 | 280 |
| Dominance Index | 0.10 | 0.12 | 0.07 |
| Shannon Index | 1.12 | 1.08 | 1.30 |
| | | | |
| *Stand 3– near middle elevational limit of woodland, relatively high proportion of very old trees (>300 yr) prior to mortality event, 39% of trees died in the event* | | | |
| Number of Structural Components | 23 | 20 | 38 |
| Number of Trees | 183 | 112 | 183 |
| Dominance Index | 0.10 | 0.17 | 0.08 |
| Shannon Index | 1.16 | 1.00 | 1.34 |

Table 3. Extreme fire senerio used in fuel models.

| FUEL CLASS | MOISTURE (%) |
|---|---|
| 1 hr | 3 |
| 10 hr | 3 |
| 100 hr | 6 |
| 1000 hr | 13 |
| live herbaceous | 32 |
| live woody | 68 |
| windspeed | 11 |

Table 4. Total percent cover, as recored on line intercept sampling, of piñon and juniper across piñon-juniper woodlands of Mesa Verde with varying degrees of mortality. Data were collected in 2010-2011.

| | | N | Mean | Stand. Dev. |
|---|---|---|---|---|
| Piñon* | No mortality | 11 | 27.1 | 28.2 |
| | low mortality | 6 | 43.6 | 9.7 |
| | high mortality | 10 | 15.9 | 11.3 |
| Juniper | none | 13 | 48.1 | 45.5 |
| | low mortality | 6 | 39.31 | 12.5 |
| | high mortality | 10 | 41.1 | 11.7 |

*F= 3.62, p=.042 for piñon; Bonferroni test shows significant difference (p=.038)between low and high Mortality levels in piñon. There was no significant difference among categories in juniper cover.

Table 5 . The degree of percent canopy closure in old-growth woodlands on Wetherill and Chapin Mesas, Mesa Verde National Park, as recorded with Hemispherical gap analysis using fisheye lens photography.

| | N | Mean | SD |
|---|---|---|---|
| No mortality | 16 | 51.6 | 24.2 |
| Low mortality | 44 | 57.2 | 11.8 |
| High mortality | 34 | 48.5 | 15.6 |
| | | | |

F=3.3, p<.05

Bonferroni Post Hoc test p<.05 for low and high mortalitys

Table 6. Fuel measurements in tons/acre, grouped by size category, across mortality categories in Mesa Verde National Park and Ute Mountain Ute Tribal Land.

| | | N | Mean | Std. Deviation |
|---|---|---|---|---|
| 1 hr fuel | No mortality | 6 | .24 | .11 |
| | Low mortality | 18 | .30 | .12 |
| | High mortality | 14 | .36 | .21 |
| 10 hr fuel | No Mortality | 6 | .80 | .46 |
| | Low mortality | 18 | 1.39 | .96 |
| | High mortality | 14 | 1. | .67 |
| 100 hr fuel | No Mortality | 6 | 1.51 | 1.4 |
| | Low mortality | 18 | 2.35 | 1.92 |
| | High mortality | 14 | 3.13 | 1.73 |
| 1000 hr sound | No Mortality | 6 | .88 | 1.12 |
| | Low mortality | 18 | 4.42 | 6.18 |
| | High mortality | 14 | 4.42 | 4.19 |
| 1000 hr rotten | No Mortality | 6 | .28 | .29 |
| | Low mortality | 18 | 1.82 | 1.41 |
| | High mortality | 14 | 1.18 | 1.16 |

**Test Statistics**

| | 1hr | 10hr | 100hr | sum 1-100hr | 1000 hr Sound | 1000 hr Rotten | Total1000 hr sound |
|---|---|---|---|---|---|---|---|
| Chi-Square | 1.575 | 4.259 | 4.294 | 5.719 | 5.578 | 8.252 | 8.205 |
| df | 2 | 2 | 2 | 2 | 2 | 2 | 2 |
| Sig. | .455 | .119 | .117 | .057 | .061 | .016 | .017 |

Table 7. Summary of results from Nexus fire models comparing fire behavior across mortality categories. These data were collected in 2010-11 from across the piñon-juniper woodlands in MVNP and UMU.

| | | N | Mean | Std. Deviation | Significance |
|---|---|---|---|---|---|
| Heat per unit area (BTU/ft2) | no mortality | 4 | 1589.7 | 948.6 | ns |
| | low mortality | 18 | 1498.0 | 942.1 | |
| | high mortality | 10 | 1274.8 | 604.9 | |
| Spread rate (Ch/hr) | no mortality | 4 | 6.9 | 4.7 | ns |
| | low mortality | 18 | 10.0 | 7.42 | |
| | high M mortality | 10 | 9.1 | 4.9 | |
| Fireline intensity (BTU/ft-s) | no mortality | 4 | 226.0 | 210.3 | ns |
| | low mortality | 18 | 344.4 | 361.6 | |
| | high mortality | 10 | 244.5 | 227.3 | |
| Flame length (Ft) | healthy | 4 | 5.0 | 2.5 | ns |
| | low mortality | 18 | 6.6 | 4.5 | |
| | High mortality | 10 | 5.5 | 2.75 | |
| Torching Index (Miles/hr) | healthy | 4 | 9.4 | 7.8 | ns |
| | low mortality | 18 | 10.5 | 13.66 | |
| | high mortality | 10 | 9.1 | 7.1 | |
| Crown Index (Miles/hr) | healthy | 4 | 57.3 | 34.8 | F=5.3, p<.01 |
| | low mortality | 18 | 27.9 | 12.3 | |
| | high mortality | 10 | 33.9 | 12.8 | |

Table 8. Results of Nexus Fire Behavior output, comparing pre-mortality (data collected 1993-1995) and post mortality (data collected 2010-2011) fire behavior, Mesa Verde National Park. Data are mean ± standard deviation.

| Year | N | Heat per unit area (BTU/ft2) | Fireline intensity (BTU/ft-s) | Flame Length (Ft) | Spread Rate (Ch/hr) |
|---|---|---|---|---|---|
| 1993 | 6 | 462 ± 572 | 131 ± 248 | 2.7 ±3.3 | 6.5 ±8.8 |
| 2010 | 16 | 1617 ± 1174 | 335± 599 | 5.5 ± 3.9 | 7.2 ±8.3 |
| | | T=2.3, p=.034 | ns | ns | ns |

Table 9. Fire behavior output summaries across fire chronosequence.

| | | N | Mean | Std. Deviation | Significance |
|---|---|---|---|---|---|
| Spread rate (Ch/hr) | <25 yrs | 57 | 30.49 | 15.31 | F=32.8, p<.001 |
| | 25-200 yrs | 5 | 43.86 | 9.32 | |
| | 200-400 yrs | 16 | 20.05 | 9.48 | |
| | old-growth | 33 | 6.44 | 6.07 | |
| Heat per until area(BTU/ft2) | <25 yrs | 57 | 1831.41 | 990.38 | F=5.1, p=.002 |
| | 25-200 yrs | 5 | 2971.60 | 1111.00 | |
| | 200-400 yrs | 16 | 1274.06 | 767.78 | |
| | old-growth | 33 | 1470.06 | 897.49 | |
| Fireline intensity (BTU/ft-sec) | <25 yrs | 57 | 1206.75 | 1129.88 | F=13.7, p<.001 |
| | 25-200 yrs | 5 | 2371.20 | 1034.46 | |
| | 200-400 yrs | 16 | 529.50 | 457.99 | |
| | old-growth | 33 | 241.12 | 430.134 | |
| Flame length (Ft) | <25 yrs | 57 | 10.63 | 4.70 | F=21.6, p<.001 |
| | 25-200 yrs | 5 | 15.70 | 3.18 | |
| | 200-400 yrs | 16 | 7.33 | 3.21 | |
| | old-growth | 33 | 4.64 | 2.89 | |

Table 10. Grasses used in post-fire aerial seeding in MVNP and UMU (BAER 1996, 2000a and 2000b; Floyd *et al.* 2006).

| Scientific Name | Common Name | Fire |
|---|---|---|
| *Pascopyrum smithii* | western wheat | Bircher (mesas); Pony (drainages and low canyon sides), and Long Mesa 2 |
| *Achnatherum hymenoides* | indian rice grass | Bircher (mesas and escarpments), Pony (mesas), Chapin 5, and Long Mesa 2 |
| *Elymus elymoides* | squirrel tail grass | Bircher (mesas), Pony (mesas and drainages), Chapin 5, and Long Mesa 2 |
| *Elymus trachycaulus* | slender wheatgrass | Bircher (mesas), Pony (drainages), Chapin 5, and Long Mesa 2 |
| *Koelaria cristata* | junegrass | Bircher (mesas), Chapin 5 |
| *Poa fendleriana* | muttongrass | Bircher (mesas), Pony (mesas), Chapin 5, and Long Mesa 2 |
| *Hesperostipa comata* | needle and thread grass | Bircher (mesas) |
| *Sporobolus cryptandrus* | sand dropseed | Bircher, and Pony (mesas) |
| *Bouteloua gracilis* | blue grama | Bircher, and Pony (mesas) |

Table 11. The cover (percent of 25 points in point frame) of native and invasive grasses and forbs in samples randomly placed throughout seeded and unseeded areas burned 10-14 years previously in Mesa Verde National Park. Data were transformed by arcsin square root transformation for statistical analysis. Untransformed mean and standard deviation shown.

| | | N | Mean | Std. Deviation | Significance. |
|---|---|---|---|---|---|
| invasive grass | not seeded | 54 | 9.85 | .044 | F= 4.1. p=.04 |
| | seeded | 59 | 3.80 | | |
| native grass | not seeded | 54 | 14.15 | .005 | F= 8.1, p=.005 |
| | seeded | 59 | 26.17 | | |
| native forb | not seeded | 54 | 13.93 | .464 | F=.54, p>.05 |
| | seeded | 59 | 11.25 | | |
| invasive forb | not seeded | 54 | 2.30 | .106 | F=2.6, p>.05 |
| | seeded | 59 | .34 | | |

Table 12 . The cover of cheatgrass and muskthistle in post fire treatments that were seeded within a few months of the fire (immediate treatment) and those that were seeded a year after the treatment (delayed seeding).

| | | N | Mean | Std. Deviation |
|---|---|---|---|---|
| Invasive grasses* | immediate seeded | 79 | 1.82 | 6.07 |
| | delayed seeded | 36 | 12.89 | 22.54 |
| Invasive forbs** | immediate seeded | 79 | .30 | 1.24 |
| | delayed seeded | 36 | 3.22 | 9.97 |
| Native grasses | immediate seeded | 79 | 19.04 | 21.06 |
| | delayed seeded | 36 | 21.89 | 27.13 |
| Native Forbs | immediate seeded | 79 | 12.61 | 20.31 |
| | delayed seeded | 36 | 16.00 | 19.38 |

*F= 6.39, p=.013
**F=16.146, p<.001

Table 13**.** Mean diversity of native and invasive plants (number of species in 9 $m^2$) in paired seeded and unseeded treatments in MVNP.

| Plant status | Treatment | Mean | Standard Deviation |
|---|---|---|---|
| Native | Seeded | 8.5 | 2.8 |
| Native | Unseeded | 9.2 | 4.0 |
| Invasive | Seeded | 1.8 | 1.0 |
| Invasive | Unseeded | 1.9 | 1.3 |

Table 14. Mean abundance of native and invasive plants (number per 9 $m^2$) in paired seeded and unseeded treatments in MVNP.

| Plant Status | Treatment | Mean | Standard Deviation |
|---|---|---|---|
| Native | Seeded | 198.1 | 60.8 |
| Native | Unseeded | 164.6 | 153.8 |
| Invasive | Seeded | 42.2 | 82.1 |
| Invasive | Unseeded | 113.7 | 46.7 |

Table 15. Precipitation and snow totals for Mesa Verde National Park, by month from 2003 to 2012. Courtesy of George San Miguel.

| **Precip** | 2003 | 2004 | 2005 | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 |
|---|---|---|---|---|---|---|---|---|---|---|
| Jan | 0.93 | 1.46 | 4.47 | 0.94 | 0.71 | 3.98 | 3.40 | 3.67 | 0.30 | 1.30 |
| Feb | 3.03 | 2.72 | 3.39 | 0.05 | 1.13 | 2.94 | 1.13 | 2.79 | 0.75 | 1.83 |
| March | 2.39 | 0.32 | 1.56 | 2.86 | 0.85 | 0.21 | 0.25 | 1.62 | 1.71 | 0.61 |
| April | 0.13 | 2.63 | 1.93 | 0.85 | 1.43 | 0.65 | 1.72 | 0.84 | 1.26 | |
| May | 0.32 | 0.00 | 1.06 | 0.11 | 2.85 | 0.98 | 1.14 | 0.14 | 1.98 | |
| June | 0.22 | 0.15 | 0.38 | 0.87 | 0.26 | 0.38 | 1.09 | 0.06 | 0.00 | |
| July | 0.15 | 1.08 | 0.80 | 1.48 | 2.46 | 2.92 | 0.49 | 2.44 | 1.61 | |
| August | 2.10 | 0.37 | 1.86 | 1.81 | 3.27 | 1.57 | 1.90 | 2.32 | 2.45 | |
| Sept. | 2.50 | 3.58 | 3.46 | 2.49 | 1.82 | 1.01 | 0.76 | 1.31 | 1.58 | |
| Oct. | 0.69 | 1.78 | 1.45 | 4.10 | 1.99 | 0.69 | 0.75 | 2.26 | 2.33 | |
| Nov. | 1.06 | 1.57 | 0.05 | 0.42 | 0.31 | 1.64 | 0.45 | 1.03 | 1.18 | |
| Dec. | 0.91 | 0.92 | 0.17 | 0.92 | 4.16 | 4.10 | 2.32 | 3.73 | 0.58 | |
| Sum | 14.43 | 16.58 | 20.58 | 16.90 | 21.24 | 21.07 | 15.40 | 22.21 | 15.73 | |
| | | | | | | | | | | |
| **Snow** | 2003 | 2004 | 2005 | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 |
| Jan | 5.20 | 14.10 | 25.50 | 10.30 | 11.30 | 32.70 | 6.80 | 35.60 | 0.30 | 14.00 |
| Feb | 25.50 | 30.30 | 13.00 | 0.80 | 4.90 | 32.40 | 12.00 | 23.50 | 8.50 | 22.50 |
| March | 17.00 | 3.80 | 5.20 | 34.00 | 3.30 | 3.30 | 1.20 | 9.30 | 6.80 | |
| April | 1.00 | 9.50 | 0.50 | 0.00 | 11.00 | 4.60 | 10.70 | 4.50 | 0.50 | |
| May | 0.00 | 0.00 | 0.00 | 0.00 | 3.00 | 0.00 | 0.00 | 0.50 | 0.00 | |
| June | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | |
| July | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | |
| August | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | |
| Sept. | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | |
| Oct. | 0.00 | 1.50 | 0.00 | 2.00 | 0.00 | 0.00 | 2.50 | 0.00 | 0.00 | |
| Nov. | 0.00 | 3.50 | 0.00 | 4.00 | 3.50 | 8.50 | 2.00 | 12.20 | 3.00 | |
| Dec. | 10.00 | 1.70 | 2.50 | 1.75 | 1.45 | 41.70 | 22.40 | 28.10 | 8.90 | |
| Sum | 58.70 | 64.40 | 46.70 | 52.85 | 38.45 | 123.20 | 57.60 | 113.70 | 28.00 | |

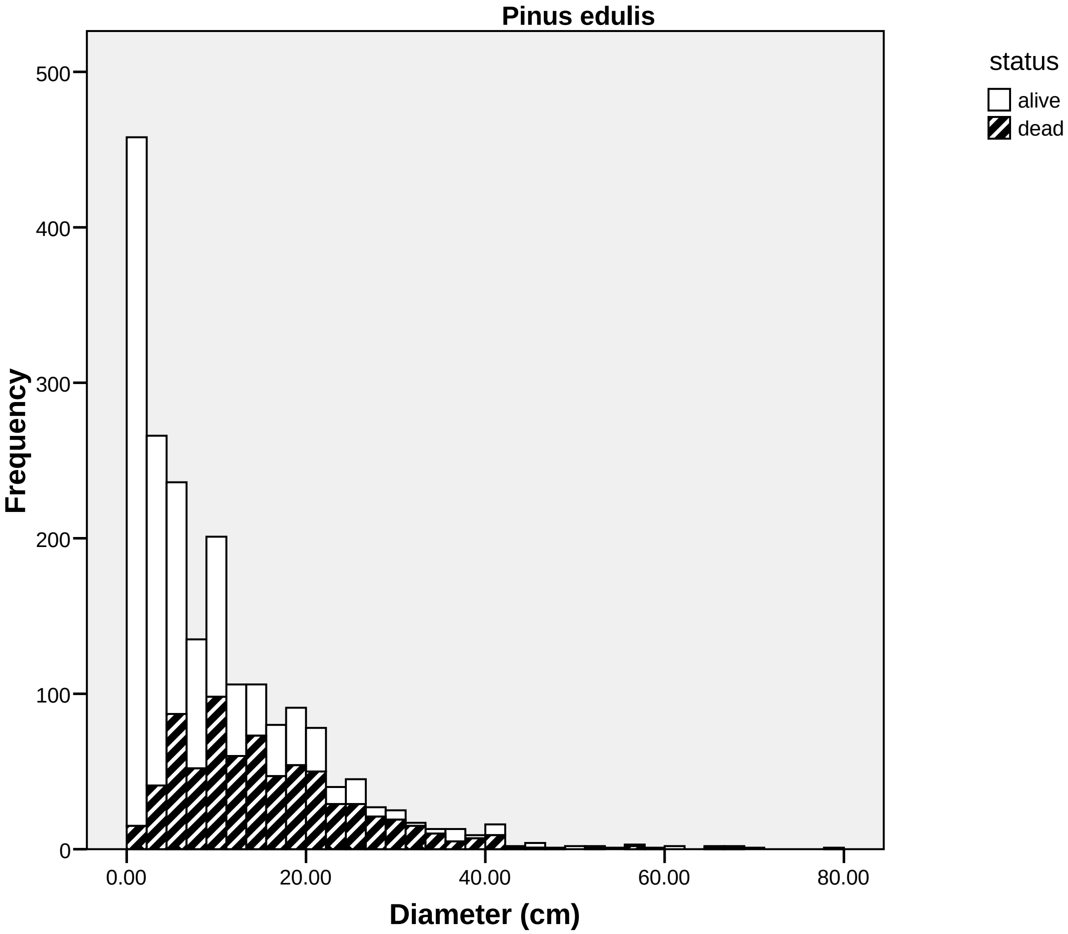


Figure 1: Patterns of mortality in Mesa Verde National Park and the adjacent Ute Mountain Ute Tribal Park following drought and beetle mortality in 2002-2004.

Figure 1a. The distribution of live and dead *Pinus edulis* in 2005.

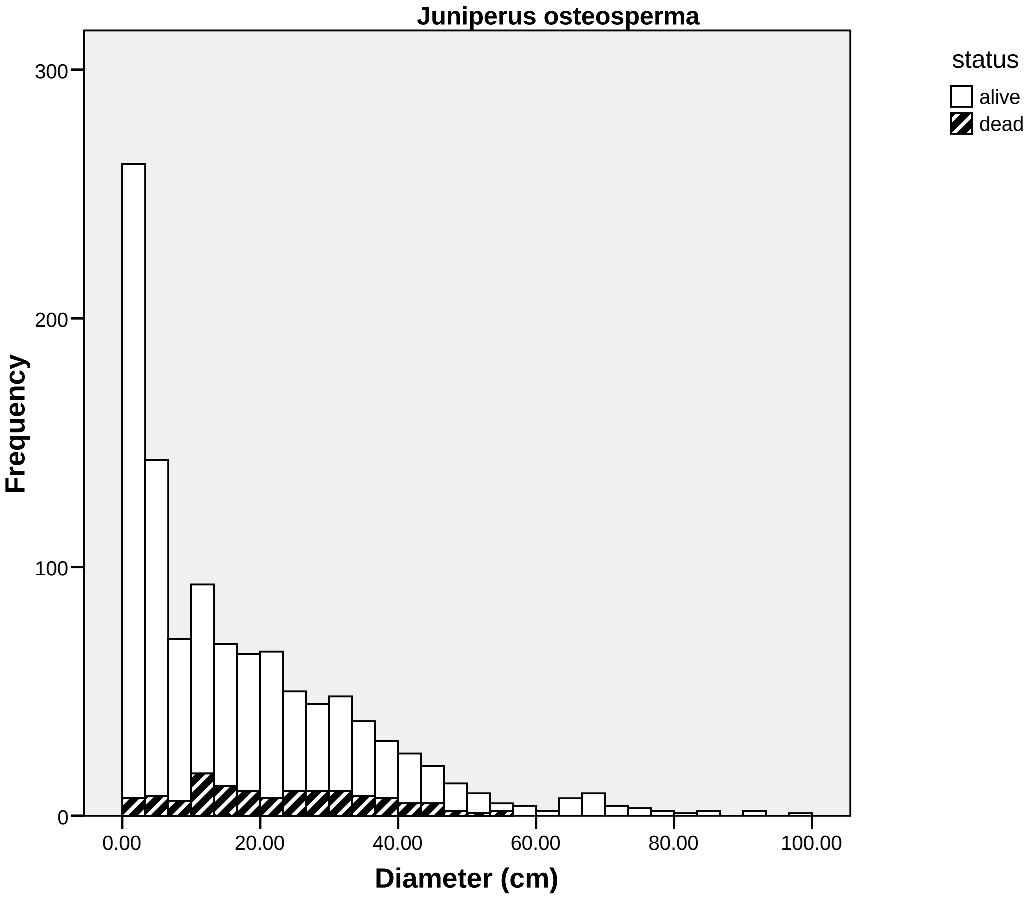
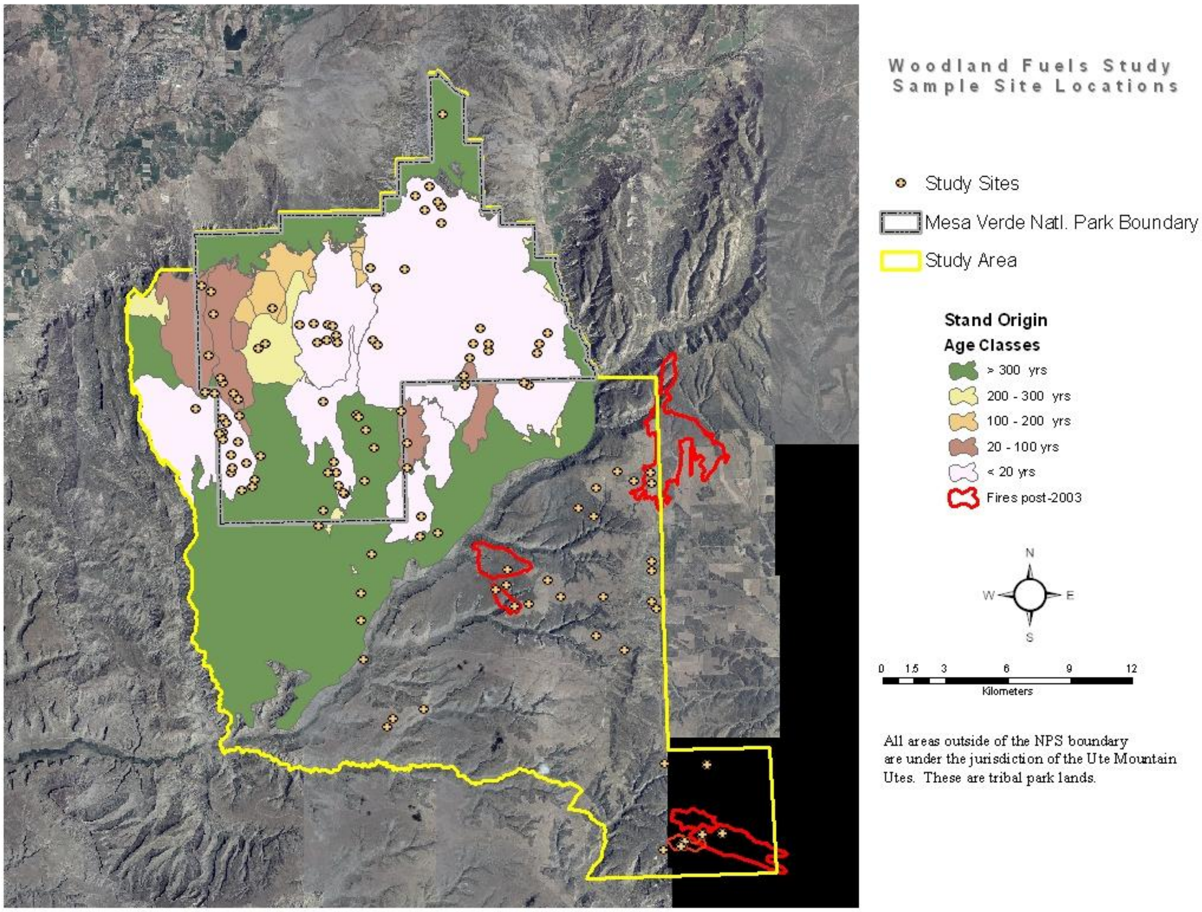


Figure 1b. The distribution of live and dead *Juniperus osteosperma* in 2005.



All areas outside of the NPS boundary are under the jurisdiction of the Ute Mountain Utes. These are tribal park lands.

Figure 2: The location of sample points within a study area defined as Mesa Verde National Park and portions of the Ute Mountain Ute Tribal land, southwestern Colorado. The western edge is bounded by the west edge of Eagle Eye Mesa above the east fork of Ute Canyon, and the southern boundary is defined by Grass Canyon. The eastern boundaries are of Mesa Verde National Park (Mancos River) and Ute Mountain Ute Tribal Park.

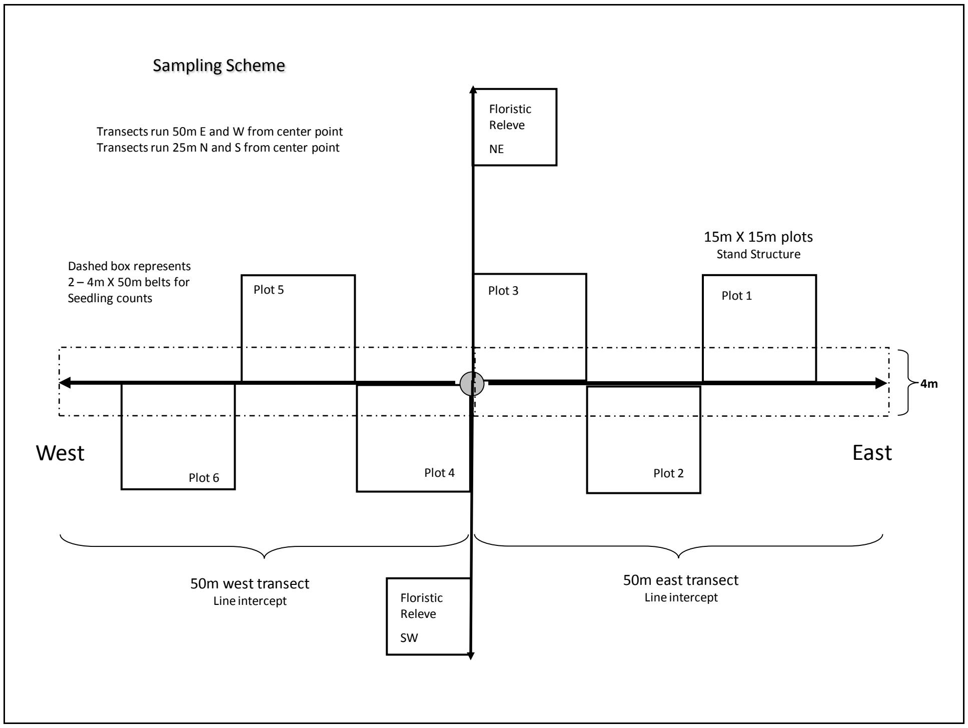


Figure 3: Arrangement of plots at each of the 115 sampling points, Mesa Verde National Park and Ute Mountain Ute Tribal Land.

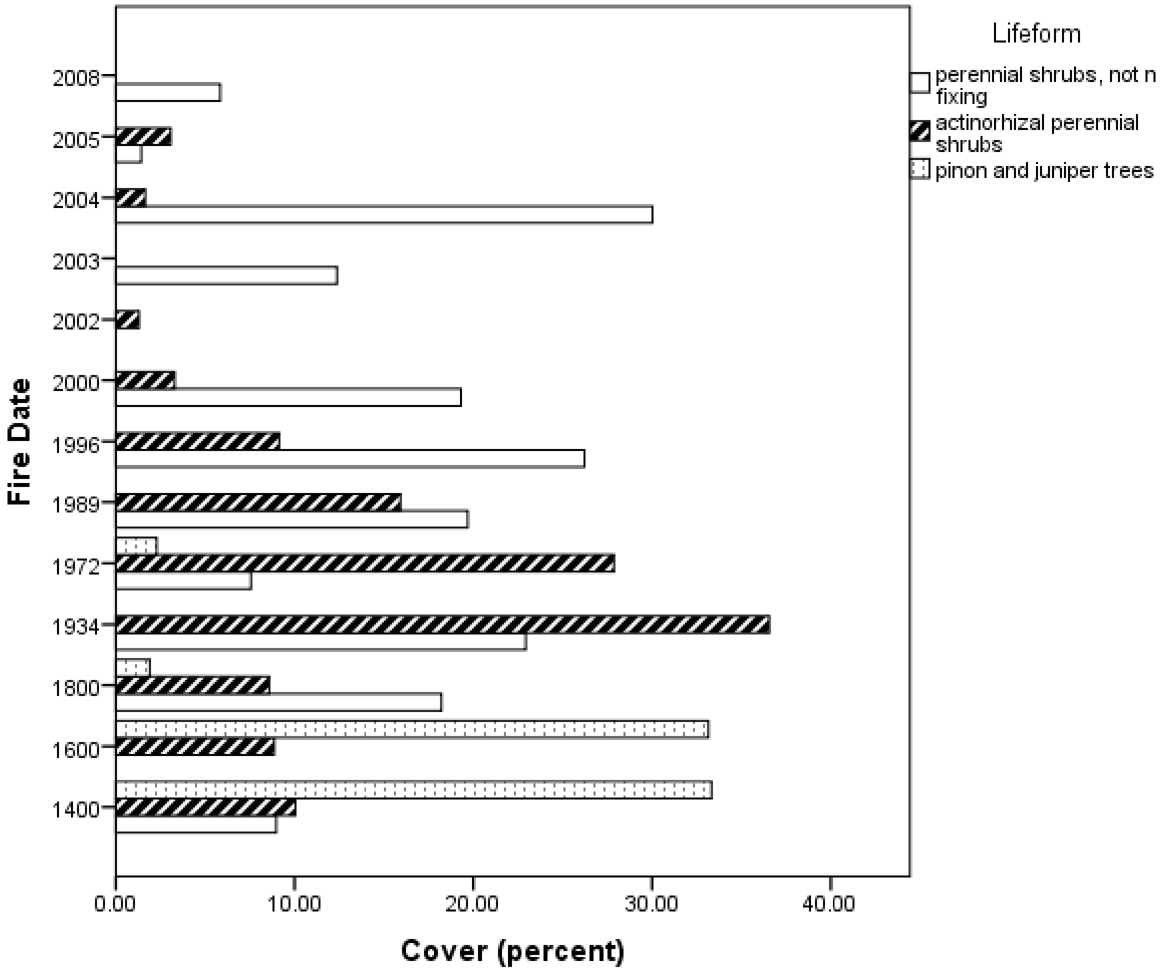


Figure 4. Patterns of tree and shrub cover across fire chronosequence, Mesa Verde National Park and Ute Mountain Ute Tribal Land.

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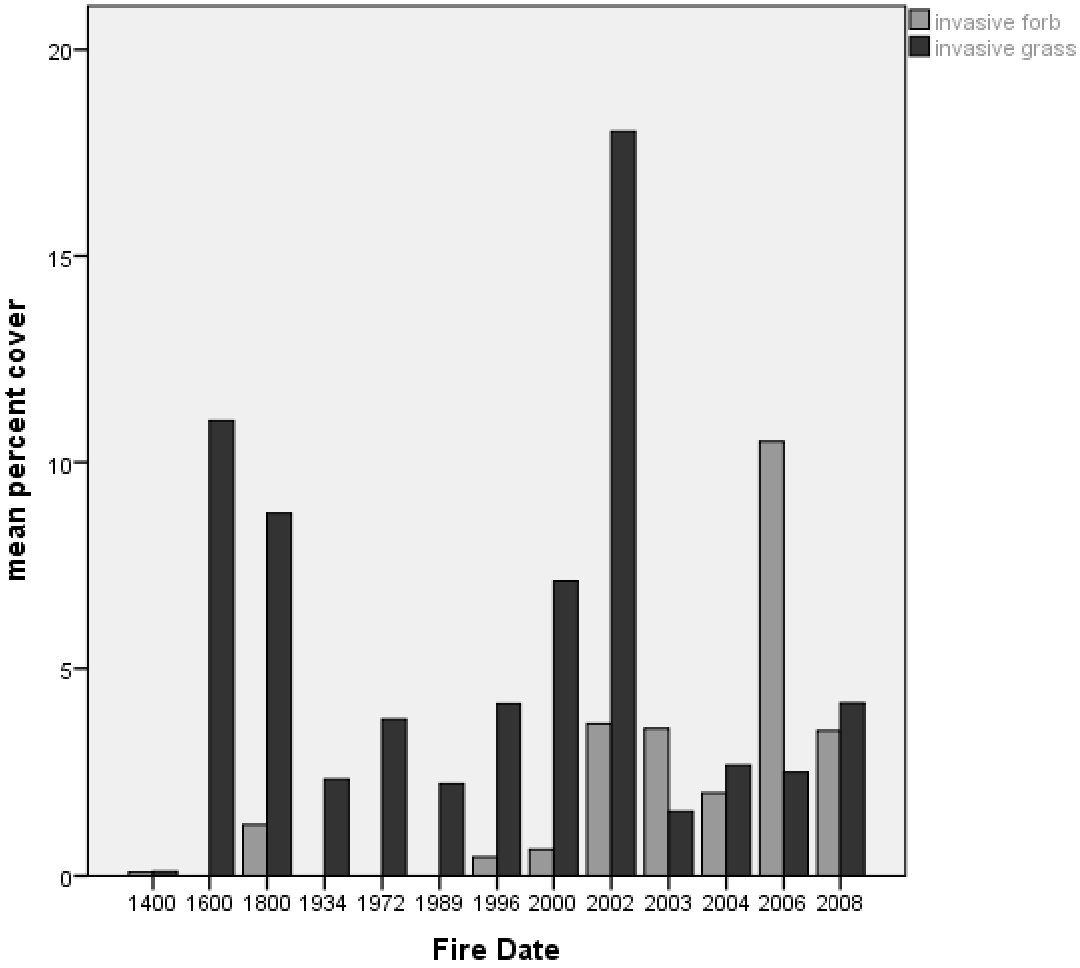

Figure 5. Percent cover of invasive grasses (*Bromus tectorum* and *Bromus inermis*) and invasive forbs (*Carduus nutans*) across the fire chronosequence.