Efficacy of resource objective wildfires for restoration of ponderosa pine (*Pinus ponderosa*) forests in northern Arizona

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ABSTRACT

Current conditions in dry forests of the western United State have given rise to policy mandates for accelerated ecological restoration on U.S. National Forest System and other public lands. In southwestern ponderosa pine (*Pinus ponderosa* Laws.) forests, mechanized tree thinning and prescribed fire are common restoration treatments but are not acceptable for all sites. Currently there is much interest in managing naturally ignited fires to accomplish restoration objectives but few studies have systematically examined the efficacy of such “resource objective” wildfires for restoring historical ranges of variability (HRV). In this study we used field plots to retrospectively sample 10 resource objective fires on two national forests in northern Arizona. We used four burn severity classes identified on Monitoring Trends in Burn Severity (MTBS) maps to stratify field sampling and compared post-fire means for 12 structure and hazardous fuels attributes to HRV and guidelines for ecosystem management. Results indicated significant differences among burn severity classes in tree density, basal area, coarse wood loads, canopy cover, and canopy fuel loads. Furthermore, areas classified as moderate (M) burn severity met HRV ranges for more attributes (67%) than did other areas in other severity classes. High (H) severity was within HRV for the fewest (17%) of the 12 attributes. Restoration ranges for large snag density, tree patch density and maximum patch size, and tree diameter distribution were not met within any burn severity class. Resource objective fire landscapes were comprised mainly (85%) of areas classified as unburned/low (U/L) and low (L) burn severity, whereas the M severity class made up just 12% of fire landscapes on average. Overall effectiveness of resource objective fires for meeting restoration objectives was 42%. Results suggested that effectiveness may be increased by managing for proportionally more moderate burn severity on these landscapes. For this, managers will be required to accept greater risk in terms of escaped fires

and high-severity fire, which, in turn, may be achieved through increasing public awareness of the potential benefits of managing wildfires for restoring ponderosa pine forest ecosystems.

Keywords: restoration objectives; historical range of variability; forest structure; hazardous fuels; forest fire management

1. Introduction

Landscape-scale changes in ecosystem structure and function resulting from intensive livestock grazing and interruption of frequent surface fire regimes have put dry forests of western North America at risk for large, severe wildfire and further degradation with changing climatic conditions (Covington et al., 1994; Moore et al., 1999; Savage and Mast, 2005; Westerling et al., 2006; Reynolds et al., 2013). Past land uses and fire exclusion have led to abnormally dense stands of trees, many of which have young cohorts that create ladder fuels, with high surface and canopy fuel loads (Fulé et al., 1997; Agee and Skinner, 2005; Reynolds et al., 2013). In addition, fine- and mid-scale spatial patterns such as tree point patterns, patch density, and patch sizes have been altered since fire regime interruption (Sánchez Meador et al., 2009). Current conditions in these forests have given rise to policy mandates for accelerated ecological restoration on US National Forest System and other public lands (Vilsack, 2009; Franklin and Johnson, 2012; USDA Forest Service, 2012; USDA, 2016). Currently, objectives under ecosystem restoration policy call for reestablishing natural ranges of variability for structure, pattern, and ecological processes to facilitate sustainability and resilience (USDA, 2016). Natural, or historical, ranges of variability (HRV) characterize prevalent states and dynamics prior to widespread anthropogenic alteration of an ecosystem (Morgan et al., 1994; Landres et

al., 1999). Generally, treatment approaches that result in structural and compositional patterns more similar to HRV are thought to provide the best chances for recovery from degradation and improvement in ecosystem function (SER, 2004; Palmer et al., 2006). Thus, in addition to informing the development of treatment prescriptions, HRV can be used as a baseline for evaluating effectiveness of restoration treatments.

In southwestern ponderosa pine (*Pinus ponderosa* Laws.) forests, mechanized tree thinning and application of prescribed fire are common approaches to restoration (Friederici, 2003). These treatments can be implemented precisely – individual trees can be selected for cutting and prescribed fire behavior can be carefully managed – and, as a result, they are known to be generally effective for accomplishing restoration objectives (Covington et al., 1997; Roccaforte et al., 2010). However, thinning and prescribed fire are relatively costly, are typically time-consuming to implement, and tree thinning is not an acceptable alternative for all sites (e.g., steep slopes, remote locations, designated Wilderness areas, etc.). For these reasons, there is currently much interest in managing naturally ignited fires to accomplish restoration and hazardous fuels treatment objectives (e.g., North et al., 2015).

Since the 1970s, national policies for managing wildfires for resource benefit have gone through numerous changes (van Wagtendonk, 2007). For example, such fires have been categorized variously as “prescribed natural fires”, “wildland fire use fires”, “resource benefit fires”, and, most recently, “resource objective fires”. National fire management policy in the U.S. no longer differentiates between resource objective and suppression fires (Meyer, 2015). In fact, current directives allow objectives to change throughout the course of a wildfire and different areas within a fire may be managed for different objectives. However, wildfires managed for beneficial outcomes commonly occur when weather and fuels conditions are

favorable, and in areas where risks to values are low. In contrast, suppression responses are typically taken during periods of more extreme weather and when risk of damage to resources is high. Thus, in many cases, tactics and objectives for individual fires are pointed distinctly toward either management for resource objectives, or suppression, regardless of policy direction. Although resource objective wildfires appear to hold promise for restoration and hazardous fuels reduction, few studies have systematically examined these assumptions across a broad geographic area. Case studies and others focused on isolated sites in the Southwest have indicated that effects within fires are variable and changes to forest structure are related to burn severity (Fulé and Laughlin, 2007; Hunter et al., 2011; Higgins et al., 2015). As burn severity classifications reflect changes to aboveground biomass, higher burn severity classes (e.g., moderate to high burn severity) indicate greater changes in forest overstory structure, and therefore would be expected to move conditions closer to HRV, than lower severity classes (Miller and Thode, 2007; Hunter et al., 2011). However, due to managers' concerns of the potential for an escaped fire, and a need to protect landscape values at risk, substantial areas within resource objective wildfire perimeters can remain unburned, or burned at low severity, and thus represent minimal changes in forest structure or crown fire potential.

To investigate their efficacy for restoring HRV of structural and hazardous fuels attributes, we sampled 10 resource objective wildfires on two National Forests in northern Arizona. We asked the following research questions: 1) Are there differences in structural and hazardous fuels attributes among burn severity classes within resource objective wildfires? 2) Are structural and fuels attributes within HRV and restoration objective ranges? 3) How effective have resource objective fires been overall for meeting restoration objectives?

2. Methods

2.1 Selection of study sites

To objectively select fires for study, the following criteria were used: 1) fires occurred in ponderosa pine or pine-oak forest types on National Forests of northern Arizona; 2) fires were documented as being naturally ignited and subsequently managed to achieve resource objectives; 3) fires were greater than 405 ha in size; 3) spatial burn severity data were available (see below); 4) time since fire was 10 years or less; 5) the site had experienced no previous resource objective fire; 6) fire sites had not been recently thinned or salvaged logged. These criteria represented common site conditions on National Forest lands in the region, and also allowed us to focus on larger fires that had potential to show variation in structural patterns at a range of spatial scales (Reynolds et al., 2013). To identify fires that met criteria related to forest type, management objectives, size, and history, we used the United States Department of Agriculture (USDA) Forest Service Region 3 fire history database. We then cross-referenced these fires with those compiled in the Monitoring Trends in Burn Severity (MTBS) database (www.mtbs.gov), for which georeferenced burn severity data layers were available. MTBS is an interagency project in the United States developed to provide a national analysis of trends in burn severity for all types of wildland fire (Eidenshink et al., 2007). The selection process yielded 17 fires meeting all criteria. From these fires, we then randomly selected 10 fires for study on the Coconino and Kaibab National Forests in northern Arizona (Fig. 1).

Most fires selected (6 of 10) had burned ≤ 5 years prior to sampling, and one fire selected burned 10 years prior to sampling (Table 1). Sites ranged from 1957 to 2423 m in elevation. Average maximum summer temperature ranged 26 to 29 °C, minimum winter

Figure 1. Locations of fires selected for study on Kaibab and Coconino National Forests. Shown are fire names (a-g) and proportion of each fire in four MTBS (Monitoring Trends in Burn Severity; www.mtbs.gov) burn severity classes: Unburned/Low (U/L), Low (L), Moderate (M), and High (H).

131 **Table 1.** Land administration, site characteristics, and average weather and fuels conditions on the day of fire ignition (Fire date) for
 132 the 10 resource objective fires studied in northern Arizona.

Name	National Forest	Elevation (m)	Soil parent material	Fire date	Maximum temperature (°C)	Relative humidity (%)	Wind speed (k hr ⁻¹)	ERC	Fuel moisture (%)		Size (ha)
									1-hr	1000-hr	
Brady	Coconino	1957-2161	Basalt/cinders	7/20/09	32	29	3	36	6	9	1612
Tucker	Coconino	2122-2299	Sandstone/limestone	5/26/09	22	34	13	22	7	13	1057
Ranger	Coconino	2029-2255	Sandstone/limestone	7/17/10	33	26	13	34	4	10	864
Raptor	Coconino	2227-2323	Basalt/cinders	7/23/09	28	38	6	24	7	11	780
Real	Coconino	2053-2301	Basalt/cinders	6/1/09	21	24	10	25	4	15	624
Independence	Coconino	2086-2233	Basalt/cinders	7/4/09	28	27	6	34	4	11	555
Marteen	Kaibab	2001-2423	Basalt	8/5/08	24	64	8	14	12	13	4366
Mudersbach	Kaibab	2019-2179	Limestone	6/23/05	27	35	8	22	7	13	3206
West	Kaibab	2136-2239	Mixed sedimentary	6/8/06	25	95	3	4	20	8	851
Transfer	Kaibab	2067-2174	Limestone/sandstone	7/18/04	29	35	5	23	7	10	672

temperature ranged -3 to -7 °C, and annual precipitation ranged 495 to 664 mm (WRCC, 2016). Precipitation in the region of the study sites falls in a bimodal pattern of summer, monsoonal rain and winter snow. Prior to the monsoon season, typically beginning mid-July, spring and early summer is distinctly dry and often windy. Soils were variable across the 10 sites, but represented those typically found in dry forests of northern Arizona (Table 1). Although forests overstories were predominantly composed of ponderosa pine, Gambel oak (*Quercus gambelii* Nutt.) is also common in these ecosystems. Other species occasionally found include Douglas-fir (*Pseudotsuga menziesii* (Mirb.) Franco), white fir (*Abies concolor* (Gord. & Glend.) Lindl. ex Hildebr.), quaking aspen (*Populus tremuloides* Michx.), southwestern white pine (*P. strobiformis* Engelm.), alligator juniper (*Juniperus deppeana* Steud.), Utah juniper (*J. osteosperma* (Torr.) Little), oneseed juniper (*J. monosperma* (Engelm.) Sarg.), and pinyon pine (*P. edulis* Engelm.). Understories of these forests are composed primarily of herbaceous species and typify the ponderosa-bunchgrass ecosystem (Brown, 1994).

To better understand environmental conditions under which the fires had been allowed to burn, we used FireFamilyPlus 4.1 (Bradshaw and McCormick, 2000) to summarize fuels and weather parameters on the day of fire ignition (Table 1). Most of the fires ignited in June-July on days with average temperatures in the mid-upper 20s (°C). Relative humidity on start dates ranged 24-95%, and daily wind speeds averaged 13 k hr⁻¹ or less. Energy release quotient was above 20 for all but two fires, and fuel moistures ranged 4-20% and 8-15% for 1-hr and 1000-hr moisture timelag classes, respectively (Table 1).

2.2 Design and Sampling

To investigate effectiveness of fires in meeting restoration objectives, we established field plots within the 10 fires using a random, stratified sampling approach. Stratification was based on the spatial burn severity maps downloaded from the MTBS database, and the following burn severity classes were identified as our sampling strata: unburned/low (U/L); low (L); moderate (M); and high (H). To classify burn severity, MTBS uses a multi-temporal approach wherein differenced normalized burn ratio (dNBR) values are calculated using Landsat imagery (30-m resolution) as the difference in pre-fire and post-fire reflectance in spectral bands 4 and 7 (near-infrared and shortwave infrared spectral ranges, respectively). Resulting values of dNBR, are partitioned into discrete classes based on interpretation of thresholds apparent in the data along with evaluation of raw pre- and post-fire imagery, plot data, and analyst experience. Burn severity classes derived following this algorithm represent a range of first-order fire effects on vegetation biomass (Eidenshink et al., 2007). Although the dNBR method has been criticized for inaccuracies, particularly as related to interpretation of forest conditions within low burn severity (i.e., minimal change) areas, and when common severity thresholds are used across forest types without field verification (Miller and Thode, 2007), Cocke et al. (2005) found dNBR classification to be strongly related to ground based measures of burn severity when multiple variables in addition to tree mortality are evaluated. We chose to use MTBS burn severity classifications as strata for our sampling for the following reasons: 1) we expected burn severity classes to reflect relative changes in forest structure and therefore influence interpretation of efficacy of fires for meeting restoration objectives; 2) by pooling field samples within severity classes, our study design was robust to potential errors noted for classification of low burn severity areas; 3) our study was restricted to evaluation of previously untreated ponderosa pine forests in northern Arizona and therefore robust to potential classification errors noted above (see

Miller and Thode 2007); and 4) MTBS maps are readily available and commonly used by public forest managers in planning responses to large fires.

To stratify sampling of fires, we used a geographic information system (GIS) to randomly assign sample plot locations within the four severity classes. To assure that sample plots would be established within the intended burn severity classes in the field, we buffered plot locations using a 0.81-ha (90 x 90-m) minimum patch size. The 0.81-ha minimum patch size along with variability in spatial patterns of burn severity within fires limited the number of suitable plot locations for sampling. Thus, two to eight sample plot locations were identified within each severity class, for each fire. In the field, we used handheld global positioning systems (GPS) units to navigate to sample plot locations. At each location, a circular, fixed-area sample plot (4400 m²) was established to measure forest structure and surface fuels. Within each plot, we collected the following measurements for all live trees (≥ 1.37 m in height): 1) species; 2) diameter at breast height (dbh; 1.37 m above ground level); 3) total height; and 4) crown base height. We also recorded species, dbh, and height for all standing dead trees within each plot. Surface fuels were measured on two, 15.2-m planar transects per plot according to methods described in Brown (1974). Fuels transects were systematically oriented along the south and west cardinal directions from plot center.

2.3 Analysis

To address our main research questions, we tested for effects of burn severity on a set of structural and hazardous fuels attributes, evaluated whether fires were meeting restoration objectives for these attributes, and calculated overall efficacy of resource objective fires for meeting restoration objectives. We first identified ranges of values to represent restoration

objectives for a set of 12 structural and hazardous fuels attributes (Table 2). These ranges were obtained from published literature concerning restoration and ecosystem management of southwestern ponderosa pine forests. Because restoration relies on historical ranges of variability (HRV) to guide development of treatment objectives (Moore et al., 1999; Sánchez Meador et al., 2011; Reynolds et al., 2013), we used published HRV information where possible to identify attribute value ranges that served as restoration objectives. For coarse wood load and large snag density, HRV information was not available; in these cases, we used published guidelines for ecosystem management (Table 2). Next, we calculated values for each of the 12 attributes from field sample plot data and GIS spatial layers. For statistical tests, we summarized data within burn severity classes and fires to produce one attribute value per burn severity class, for each fire ($n=6-10$). Attribute values calculated from sample plot data were tree density (no. ha^{-1}), basal area ($\text{m}^2 \text{ ha}^{-1}$), coarse wood load (Mg ha^{-1}), large snag density (no. ha^{-1}), canopy base height (m), canopy fuel load (kg m^{-2}), and canopy bulk density (kg m^{-3}). Coarse wood load was calculated from planar transect data according to methods in Brown (1974) for surface fuels >7.6 cm in diameter. Large snags were defined as standing dead trees >45.7 cm dbh, and >9.2 m in height (Ganey et al., 2015). Canopy bulk density and canopy fuel load were calculated using allometric equations relating dbh to component biomass (i.e., foliage and fine branches) for individual tree species and followed methods outlined by Roccaforte et al. (2015).

Canopy cover (%) and spatial metrics based on canopy cover (tree patch density, mean tree patch size, and maximum tree patch size) were derived for each fire using National Agricultural Imagery Program (NAIP) imagery (1 x 1 m resolution) acquired in 2015 (www.usgs.gov). Imagery was loaded into a GIS and a ratio band was created by dividing the near infrared and red (bands 4 and 1, respectively) (Pelz and Dickinson, 2014). The

224 **Table 2.** Attributes and ranges of values used to evaluate efficacy of resource objective fires for restoring ponderosa pine forest
 225 structure in northern Arizona.

Attribute	Objective range	Source
Tree density (no. ha ⁻¹)	56 – 138*	Reynolds et al. 2013
Basal area (m ² ha ⁻¹)	9 - 17*	Reynolds et al. 2013
Tree diameter distribution	q [†] = 1.2	Reynolds et al. 2013
Coarse wood (Mg ha ⁻¹)	11 - 45 ¹	Brown et al. 2003
Large snag density [£] (no. ha ⁻¹)	4.9 ¹	Lowe 2006; Ganey et al. 2015
Tree patch density (no. ha ⁻¹)	10 - 27	Sánchez Meador et al. 2011
Mean tree patch size (ha)	0.004 - 0.08	Sánchez Meador and Moore 2010; Sánchez Meador et al. 2011
Maximum tree patch size (ha)	0.013 - 0.290	White 1985; Sánchez Meador et al. 2011
Canopy cover (%)	14.0-19.6*	Reynolds et al. 2013
Canopy base height (m)	5 -7	Roccaforte et al. 2015; Fulé et al. 2002
Canopy fuel load (kg m ⁻²)	0.22 - 0.51	Fulé et al. 2002; Roccaforte et al. 2008; Roccaforte et al. 2015
Canopy bulk density (kg m ⁻³)	0.010 - 0.034	Fulé et al. 2002; Fulé et al. 2004; Roccaforte et al. 2008; Roccaforte et al. 2015

226 * Interquartile range

227 † Basal area – maximum diameter – diminution quotient (Bdq)

228 ¹ Ecosystem management guidelines; lack of HRV information

229 £ Large snags were defined as > 45.7 cm dbh, and > 9.2 m height

ratio band was merged with the red, green, and blue bands using ERDAS Imagine software, version 16.0. Test classifications indicated that 25 classes limited confusion between tree canopy and forest openings. Thus, a 25-class unsupervised classification was run and then each class was evaluated by overlaying with NAIP imagery to verify canopy, no canopy, confused, or canopy shadow classes. Google Earth and the original NAIP imagery were used to verify the classification. Next, confused and canopy shadow classes were used to mask the NAIP imagery, and a 12-class unsupervised classification was run on the masked image. Twelve classes proved most effective in distinguishing between tree canopies and openings although no formal accuracy assessment was performed on this or any of the other classifications. Again, each individual class was evaluated against the NAIP imagery and given the appropriate “canopy” or “no-canopy” label. On fires that had a shrub component, a third mask/classification/evaluation routine was run to separate shrub cover from tree canopies. The resulting “canopy”/“no-canopy” classifications were combined, and percentages of focal areas classified as canopy cover were calculated for each burn severity class polygon (i.e., MTBS maps) sampled with field plots. To calculate tree patch metrics, distinct tree canopy patches (continuous areas classified as canopy cover and thus representing interlocking tree canopies) were filtered to include only those patches greater than 100 m², thus minimizing the inclusion of single trees. Once filtered, patch metrics were quantified and summarized by MTBS severity class for each fire. Tree patch density, mean tree patch size, and maximum tree patch size were estimated from the canopy cover layers using the *raster* (Hijmand and Etten, 2012) *rgdal* (Bivand et al., 2015) and *SDMTools* (VanDerWal et al., 2014) packages in R version 3.2.1 (R Core Team 2015). As with canopy cover, patch metric estimates were generated for the burn severity polygons that were sampled in the field.

We used analysis of variance (ANOVA) (Kuehl, 1994) to analyze for effects of burn severity class on forest structure and hazardous fuels attributes. When main effects were found ($p \leq 0.05$) we used the Tukey-Kramer HSD (honestly significant difference) post-hoc tests to compare means among the four burn severity classes (Kuehl, 1994). To test whether restoration objectives were being met within each of the four burn severity classes, we calculated 95% confidence intervals (CI; $df = 5-9$) for each attribute of interest. CIs were compared with restoration objective ranges and when CIs overlapped with the objective ranges we considered the objective to be met. Targets were considered not met if CIs did not overlap with reference ranges, or if the CI included zero. We used chi-squared tests ($df = 2$) to compare live tree diameter distributions within burn severity classes sampled in field plots to theoretical distributions produced using the basal area-maximum diameter method (BDq) and a diminution quotient (q) of 1.2 (Reynolds et al., 2013). Distributions were based on three diameter classes: 0-20 cm; 20.1-40 cm, and >40 cm dbh. “Effectiveness” (%) of resource objective fires for restoring forest structure was defined as the percentage of the 12 structural and hazardous fuels attributes where objective ranges were met. We evaluated effectiveness in this manner for each of the four burn severity classes. “Area-weighted effectiveness” for each burn severity class was calculated as effectiveness * “landscape representation”. “Landscape representation” was the mean percentage of fire areas comprised by each burn severity class. Lastly, “overall effectiveness” was calculated as the sum of the area-weighted effectiveness values for each burn severity class.

3. Results

3.1 Burn severity effects

Mean landscape representation (i.e., fire area (%)) within each severity class) of the four burn severity classes were 35.8%, 50.0%, 12.0%, and 1.9% for U/L (unburned/low), L (low), M (moderate), and H (high), respectively (Fig. 2). Clear trends in attribute means were apparent across burn severity classes, and six of the 12 restoration and hazardous fuels attributes were significantly ($p < 0.05$) affected by burn severity (Table 3). For example, tree density was significantly higher in the U/L and L severity classes compared with the M class, which in turn showed higher density than the H class. Compared with the U/L class, fire-related mortality reduced tree density by 31.2%, 64.5%, and 97.5% on average in L, M, and H classes, respectively. Basal area followed the same pattern as tree density with significantly higher values in U/L and L classes, and the lowest mean BA value in the H class (Table 3). On average, BA was reduced by fire by 11.4%, 51.9%, and 98.3% in L, M, and H classes, respectively, compared with BA in the U/L class. Coarse wood load was significantly lower in U/L and L classes than M and H severities, which showed statistically similar amounts of coarse wood. We found no main effects for large snags, and densities were similar among the burn severity classes. Similarly, there were no effects of severity on tree patch density, mean tree patch size, or maximum tree patch size (Table 3). Canopy cover was significantly higher in U/L, L, and M classes compared with the H severity class. There were no effects of burn severity on canopy base height. Canopy fuel load was significantly affected by burn severity with a clear trend of decrease from the U/L to the H severity class (Table 3). Canopy bulk density showed a similar pattern as canopy fuel load with significantly decreasing amounts with increasing burn severity.

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295 **Figure 2.** Effectiveness (%) of burn severity classes (Unburned/Low (U/L), Low (L), Moderate
 296 (M), and High (H)) for meeting restoration and hazardous fuels target ranges, and mean
 297 landscape representation (fire area (%) within each burn severity class) for 10 resource objective
 298 fires studied in ponderosa pine forests of northern Arizona. Overall effectiveness (area-weighted
 299 for all severity classes) was 42%.

300 **Table 3.** Means* (and 95% CIs) for structural and hazardous fuels attributes evaluated within burn severity classes (see text for
301 definitions). Gray shading indicates confidence intervals that overlap objective ranges (see Table 2), whereas no shading indicates
302 attributes for which restoration objectives were not met.

Attribute	Burn severity			
	U/L	L	M	H
Tree density (no. ha ⁻¹)	485.5 (166.0)a	333.8 (125.8)a	172.2 (97.1)b	12.0 (12.1)c
Basal area (m ² ha ⁻¹)	23.5 (5.8)a	20.8 (7.0)a	11.3 (5.3)b	0.4 (0.5)c
Coarse wood (Mg ha ⁻¹)	10.7 (8.4)a	12.2 (5.9)a	29.6 (13.1)b	45.1 (27.7)b
Large snag density (no. ha ⁻¹)	1.0 (1.7)	2.2 (2.4)	5.9 (6.5)	9.4 (13.6)
Tree patch density (no. ha ⁻¹)	67.4 (18.2)	61.6 (14.2)	78.3 (19.4)	77.8 (37.0)
Mean tree patch size (ha)	0.008 (0.005)	0.008 (0.004)	0.005 (0.002)	0.002 (0.001)
Maximum tree patch size (ha)	13.0 (10.4)	20.0 (17.4)	6.4 (10.8)	1.1 (1.3)
Canopy cover (%)	41.1 (7.1)a	44.3 (8.3)a	31.9 (12.5)a	13.7 (14.2)b
Canopy base height (m)	3.6 (1.0)	4.6 (1.3)	6.0 (2.9)	3.2 (4.8)
Canopy fuel load (kg m ⁻²)	0.687 (0.182)a	0.579 (0.196)ab	0.311 (0.150)bc	0.012 (0.019)c
Canopy bulk density (kg m ⁻³)	0.050 (0.013)a	0.038 (0.012)ab	0.024 (0.012)b	0.002 (0.003)c

303 * Different lowercase letters denote statistically different means among burn severity classes at $p < 0.05$.

3.2 Restoration objectives

Restoration objectives for structural and hazardous fuels attributes were met to various degrees by all burn severity classes (Table 3). Tree densities were higher than HRV in the U/L (unburned/low) and L (low) severity classes. Tree density in the M (moderate) severity class fell within the HRV objective range, but was variable in the H (high) severity class (CI included zero) (Table 3). Class U/L was above HRV for basal area, whereas BA in L and M classes was within HRV. BA in the H burn severity was again too variable. All burn severity classes were within the objective range for coarse wood load (Table 3). Large snag densities were highly variable, and CIs for all burn severity classes included zero (Table 3). HRV for tree patch density was not met by any burn severity classes. In contrast, HRV for mean patch size was met within all burn severity classes (Table 3). Maximum tree patch size was not met within U/L and L classes, whereas CIs within M and H severity classed included zero. Canopy cover means in the U/L and L burn severity classes were above HRV. Only the M severity class was within HRV for canopy cover (Table 3). Canopy base height was below HRV in the U/L class, within the objective range in the L and M severity classes, and CI for canopy base height in the H severity class included zero. HRV for canopy fuel load was met by all burn severity classes except H (Table 3). Canopy bulk density was above HRV in the U/L class, whereas both L and M severity classes were within HRV. Again, the CI for canopy bulk density in the H severity class included zero. No burn severity classes met the objective for tree diameter distribution; U/L, L, and H classes were deficient in larger trees (> 40.0 cm dbh), and U/L, L, and M severity classes showed an overabundance in mid-sized trees (20.1 – 40.0 cm dbh) (Fig. 3).

Figure 3. Proportion of trees density (and stand errors) in three tree diameter classes for each of the four burn severity classes: Unburned/Low (U/L), Low (L), Moderate (M), and High (H). Gray shading indicates theoretical historical range of variability as determined using the basal area-maximum diameter method and diminution quotient of 1.2 (Reynolds et al. 2013). Diameter distributions for all burn severity classes were significantly different ($p < 0.05$) from HRV.

In terms of effectiveness, the U/L burn severity class met restoration objectives for 25% of the attributes and the L severity class was 50% effective. The M class met more restoration targets than the other classes and was 67% effective. Burn severity class H was least effective as it met objectives for just 17% of the attributes examined. Overall effectiveness (area-weighted for all severities) of resource objective fires for restoring ponderosa pine structure and fuels characteristics was 42% (Fig. 2).

4. Discussion

4.1 Efficacy of Resource Objective Fires

Random sampling of 10, large (> 405 ha) resource objective wildfires on the Coconino and Kaibab National Forests indicated that these fires have been variably effective for restoration of stand structure and hazardous fuels characteristics in ponderosa pine forests of northern Arizona. Effectiveness was related to burn severity and we observed clear trends among the four burn severity classes in nearly all structural and hazardous fuels attributes. Means were statistically different among burn severity classes for 50% of the attributes examined. These findings suggest that before fires occurred, forests were relatively homogenous in terms of structure and fuels. Widespread exclusion of frequent surface fires beginning in the latter part of the 19th century resulted in millions of acres of dense stand conditions in ponderosa pine forests of the American Southwest (Moore et al. 1999). Our results indicate that recent resource objective wildfires overall have only moderately reduced stand structure and hazardous fuel loading, as more than 85% of the area within the fires on average was classified as U/L (unburned/low) to L (low) burn severity.

Areas classified as M (moderate) burn severity were 33% to 300% more effective than the other severity classes at meeting restoration objectives. Further, HRV for tree density and canopy cover were only met in M severity areas (Table 3). However, areas classified as M burn severity made up just 12% of the fire landscapes on average (Fig. 2). Other studies of resource objective fires have also shown greater effectiveness of moderate-severity fire for meeting restoration and hazardous fuels objectives as compared with low-severity fire (Fulé and Laughlin, 2007; Taylor, 2010; Hunter et al., 2011; Higgins et al., 2015). For example, Hunter et al. (2011) found that moderate/high-severity areas of resource objective fires in ponderosa pine forests of the Gila National Forest in New Mexico showed lower tree density, basal area, and canopy bulk density, and higher crown base height than low-severity and unburned areas. In their study, low burn-severity areas reduced tree density and increased crown base height but showed no difference in basal area or crown bulk density compared with unburned areas (Hunter et al., 2011). In our study, areas classified as L burn severity comprised more landscape area than any other burn severity classes (Fig. 2). Similarly, Meyer (2015) found that resource objective wildfires in frequent-fire forest types of the southern Sierra Nevada were dominated by low-severity fire. In this sense, the fires studied by Meyer (2015) were within natural ranges of variation (i.e., HRV; see Landres et al., 1999) for burn severity; however, effectiveness of these fires for restoring forest structure was not examined. We found L-severity areas to be 50% effective at meeting restoration objectives; however, L burn severity met no structural or fuels attributes uniquely. Objective ranges for all attributes met by L were also met by M burn severity, and 50% (3 of 6) of those met by L (low) were also met by the U/L severity class. Higgins et al. (2015) found that a low-severity, single-entry resource objective fire in a white-fir-encroached ponderosa pine forest of Grand Canyon National Park did little to restore tree species

composition, but second-entry fires reduced white fir regeneration. It is well known that low-severity prescribed fire meets few restoration objectives in these ecosystems if not preceded by mechanized tree thinning (Sackett et al., 1996; Fulé et al., 2002; Hunter et al., 2011; Roccaforte et al., 2015).

Areas classified as H severity showed highly variable mean intervals, and intervals for two-thirds of the attributes we examined included zero (Table 3). Further, CIs for tree density, basal area, canopy cover, canopy fuel load, and canopy bulk density in the H severity class were all lower than HRV. These results indicate that quantities of structural elements such as live trees, snags, and logs were very low, and often completely absent in H-severity areas. Historically, fine-grained forest structural patterns were maintained by frequent surface fire (Swetnam and Baisan, 1996; Van Horne and Fulé, 2006; Abella and Denton, 2009; Sánchez Meador et al., 2011), and forest openings are thought to have been generally <0.20 ha in size (Pearson, 1923). There is very little documented evidence of larger high-severity patches in presettlement ponderosa pine forests of the Southwest (but see Iniguez et al., 2009). Although Williams and Baker (2012) surmised that large landscapes of ponderosa pine and dry mixed conifer forests in northern Arizona experienced historical high-severity fire, these conclusions, and findings based on similar assumptions, have been sharply criticized and are not supported by dendrochronological analyses (Fulé et al., 2013; Huffman et al., 2015; Stephens et al., 2015; Stevens et al., 2016). In fact, historical ranges of variability for patch metrics (e.g., tree patch density, mean patch size, and maximum patch size) based on dendrochronological reconstructions suggest that frequent surface fires in ponderosa pine forests of the Southwest generally maintained fine-grained structural patterns, and large openings created by infrequent, high-severity fire were rare (Sánchez Meador et al., 2011). Similarly, stand reconstructions at

several northern Arizona sites have indicated historical “inverse-J”-shaped diameter or irregular, uneven-aged distributions, which further supports the hypothesis that stand replacing fires were very uncommon prior to Euro-American settlement of the region (Fulé et al. 1997, 2002, 2006, Sánchez Meador et al., 2010). Thus, the H burn severity class was least effective at meeting structural and hazardous fuels objectives, and conditions within these areas appeared to be less desirable even than those within U/L areas. Indeed, H-severity areas showed those conditions resulting from uncharacteristically severe fire that restoration prescriptions are commonly developed to mitigate (Reynolds et al., 2013).

Areas classified as U/L were 25% effective and comprised 36% of fire landscapes on average (Fig. 2). Although fire effects on forest structure were nonexistent to minimal, these areas surprisingly fell within restoration objective ranges for three attributes – coarse wood load, mean tree patch size, and canopy fuel load (Table 3). This likely resulted, in part, from broad ranges for these attributes. For example, Brown et al. (2003) suggested an optimal quantity of coarse woody material to be 11-45 Mg ha⁻¹ after fire in western dry forests, and this broad range was intended to balance acceptable fire hazard and potential severity with needs for soil productivity as well as wildlife habitat. We adopted this range as an objective in our study as very little is known concerning historical amounts of coarse wood and hence HRV information is not readily available. Other studies in southwestern ponderosa pine forests have reported background (i.e., within untreated areas) coarse wood loads to range between 11 and 45 Mg ha⁻¹ (Fulé et al., 2002; 2006). Similar to coarse wood load, relatively little research has quantified historical ranges for spatial metrics such as patch density or patch size (Reynolds et al., 2013). We found a clear pattern of decreasing mean patch size with increasing burn severity (Table 3); however, the reported HRV for patch size spanned more than two orders of magnitude (Table 2).

Thus, although large dead and down logs and tree spatial patterns are important for wildlife habitat quality and ecosystem function (Ganey et al., 2015; Rodman et al., 2016), our results indicate that these attributes are relatively poor indicators (i.e., too variable or insensitive) of fire effectiveness or restoration success in southwestern ponderosas pine ecosystems. Canopy fuel load in the U/L burn severity class was within the restoration objective range and low (0.687 kg m^{-2}) compared with other reported values for undisturbed ponderosa pine forests. Canopy fuel loads commonly range between 0.7 and 1.2 kg m^{-2} in untreated areas where the historical fire regime has been disrupted (Fulé et al., 2001; 2002, Roccaforte et al., 2008; 2015). Canopy fuel load is important in affecting potential fire behavior, and although this attribute in U/L areas was within HRV, we found it to be sensitive to burn severity (Table 3) and suggest that it is valuable as an indicator to evaluate resource objective fire efficacy.

4.2 Limitations

In our study, we evaluated resource objective wildfires for meeting restoration objective ranges for forest structure and hazardous fuels; however, HRV information was either scant or not available for attributes related to coarse woody material (logs and large snags) and tree spatial patterns. We chose to include these attributes in our analysis due to their importance in influencing ecological function and because they are commonly considered in ponderosa pine restoration frameworks (e.g., Reynolds et al., 2013). More work is needed to accurately describe coarse wood quantities and spatial characteristics of trees and tree groups, perhaps by utilizing precise stem mapping and dendrochronological reconstruction techniques (see Sánchez Meador et al., 2011).

In our study we could not examine the importance of pre-fire forest conditions in affecting burn severity patterns or resulting structural characteristics. However, clear directional trends in attribute means among burn severity classes and relatively low variability in U/L and L classes suggest relatively homogenous pre-fire structural conditions. Thus, dNBR classification (MTBS) appeared to be adequate for coarse assessments of post-fire effects and stratifying sampling to evaluate efficacy of resource objective fires for restoring ponderosa pine forests in northern Arizona. Dense forest conditions resulting from historical fire exclusion across large landscapes in the Southwest have been widely discussed and are the basis for policy mandates to accelerate restoration treatments (Covington et al., 1994; Fulé et al., 1997; Moore et al., 1999; Reynolds et al., 2013).

In our study, we did not evaluate undesirable effects that may be associated with different burn severity classes. For example, other studies have reported expansion of populations and increases in cover of exotic plant species after high severity fire in western dry forests (Crawford et al., 2001; Rew and Johnson, 2010; Abella and Fornwalt, 2015), although these patterns are inconsistent (Dodge et al., 2008; Kuenzi et al., 2008). Other important undesirable effects that may be related to burn severity but that we did not address in our study include insect and disease responses, impacts on soils, and hydrologic changes. Lastly, we focused our work on single-entry fires and did not study possible additive effects of multiple burns. Based on our observations, single-entry fires are more common than multiple-entry sites on USDA Forest Service landscapes in the Southwest, with the possible exception of some designated Wilderness areas. Holden et al. (2007) found that multiple entries with resource objective wildfires reduced density of small-diameter trees in two remote forest locations in Arizona and New Mexico. Hunter et al. (2011) reported that areas receiving multiple fire entries were similar to single-

entry, low-severity fires in terms of tree density, basal area, and canopy bulk density. More research is needed to better assess whether multiple-entry fires result in greater restoration benefits than single-entry fires, particularly in the low burn severity areas (i.e., repeated low-severity fire) that dominate these fire landscapes.

4.3 Implications

National Forest System policy mandates accelerated ecological restoration treatments, and for frequent-fire ponderosa pine forests, restoration of overstory structural patterns more similar to HRV can be consistent with improving resistance to increased drought and fire occurrence (Fulé, 2008; Reynolds et al., 2013; Stephens et al., 2013). We acknowledge that the use of HRV to set structural and hazardous fuels treatment objectives may not be feasible on many sites due to a variety of social, economic, and logistical limitations (Higgs, 1997; Reynolds et al., 2013), and indeed, several authors have questioned the continued focus on precise approximation of HRV, particularly in light of uncertainties associated with climatic change (Falk, 2006; Millar et al., 2007; Hart et al., 2015; Jacobs et al., 2015). Concerns related to climate change as well as a desire to incorporate contemporary societal values have led some authors to suggest “future ranges of variation” (FRV) as an alternative concept for setting restoration objectives (Duncan et al. 2010). However, simulation of long-term treatment responses with a warming climate suggest that sustaining dry forests in the Southwest is tenuous under more severe climate change scenarios, but restoration treatments that reduce stand density and reintroduce surface fire may lower tree mortality and extend persistence of dominant species such as ponderosa pine (Shive et al., 2014; Tarancón et al., 2014; Stoddard et al., 2015). Similarly, resource objective wildfires in these systems may meet a greater number of restoration objectives and increase ecological resilience if managed to allow more moderate-severity

burning to occur. Our comparison of conditions among burn severity classes suggested that HRV objectives are met if fire-related mortality reduces tree density by about 64% and basal area by 11-52%. In contrast, tree density remained higher than HRV when fire-related mortality was 31% or less. Greater burn severity may be sought in areas where values at risk are few such as in more remote wildland zones, and by permitting fires to burn under more extreme weather conditions. Fires evaluated in our study took place under conditions of high relative humidity, low wind speed, and high fine fuel moistures (Table 1) (see Fulé et al., 2002; and Roccaforte et al., 2008). This likely resulted in combined areas of U/L and L burn severity classes occurring on over 85% of the fire landscapes. Allowing fires to burn more actively to achieve greater representation of the M severity class requires managers to accept more risk in terms of the potential for escaped fires. In addition, we found the H burn severity class to meet the fewest restoration objectives, and it is likely that landscape representation of this class will increase along with increases in M burn severity. These risks may be managed, however, through increasing public awareness of the potential ecological and economic benefits of fire use, and greater coordination among public land management agencies (North et al., 2015).

Acknowledgements

We thank staff and students of the Ecological Restoration Institute at Northern Arizona University for field data collection and logistical support. In particular, W. Chancellor, S. Curran, D. Normandin, M. Sensibaugh, and J. Springer were instrumental in assisting completion of this work. U.S. Forest Service staff from Coconino and Kaibab National Forests provided assistance in fire location and permitting. We thank A. Thode for discussions related to burn severity classification, and L. Wadleigh and Q. Johnson for feedback on project implications.

This research was supported by a grant from the USDA Forest Service. Northern Arizona University is an equal opportunity provider.

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