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Short-term ecological consequences of collaborative restoration treatments in ponderosa pine forests of Colorado

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ABSTRACT

Ecological restoration treatments are being implemented at an increasing rate in ponderosa pine and other dry conifer forests across the western United States, via the USDA Forest Service's Collaborative Forest Landscape Restoration (CFLR) program. In this program, collaborative stakeholder groups work with National Forests (NFs) to adaptively implement and monitor ecological restoration treatments intended to offset the effects of many decades of anthropogenic stressors. We initiated a novel study to expand the scope of treatment effectiveness monitoring efforts in one of the first CFLR landscapes, Colorado's Front Range. We used a Before/After/Control/Impact framework to evaluate the short-term consequences of treatments on numerous ecological properties. We collected pre-treatment and one year post-treatment data on NF and partner agencies' lands, in 66 plots distributed across seven treatment units and nearby untreated areas. Our results reflected progress toward several treatment objectives: treated areas had lower tree density and basal area, greater openness, no increase in exotic understory plants, no decrease in native understory plants, and no decrease in use by tree squirrels and ungulates. However, some findings suggested the need for adaptive modification of both treatment prescriptions and monitoring protocols: treatments did not promote heterogeneity of stand structure, and monitoring methods may not have been robust enough to detect changes in surface fuels. Our study highlights both the effective aspects of these restoration treatments, and the importance of initiating and continuing collaborative science-based monitoring to improve the outcomes of broad-scale forest restoration efforts.

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1. Introduction

Wildfires are increasing in frequency, extent, and severity throughout dry conifer forests of the western United States (US; Westerling et al., 2006; Robichaud et al., 2014; Jolly et al., 2015), highlighting a need for proactive management actions to increase social and ecological resilience to these events (North et al., 2015; Smith et al., 2016). Fuels reduction treatments have been conducted for many decades to decrease the probability of large, severe wildfires in these forests, which have been widely altered by anthropogenic land use practices and changes in climate (Cooper, 1960; Hunter et al., 2007; Stephens et al., 2012). More recently, there has been an emphasis on implementing treatments with broader ecological restoration objectives (e.g., Covington et al., 1997; Brown et al., 2001; Allen et al., 2002; Youngblood et al., 2006; Fiedler et al., 2010; Korb et al., 2012; Underhill et al., 2014; Stephens et al., 2015). Restoration treatments are designed to address the ecological degradation that has been caused by anthropogenic stressors (SER, 2004; Fulé et al., 2006); they aim to create more characteristic and disturbance-resilient conditions that are defined in terms of ecological structure and function (Moore et al., 1999; Allen et al., 2002; Larson and Churchill, 2012). Typical outcomes of such restoration treatments in dry conifer forests of the western US include reduced risk of high-severity fire across large areas, as well as a more open and heterogeneous forest structure, greater diversity and cover of native understory plants, and greater diversity of habitats for native wildlife species (e.g., Reynolds et al., 2013; Hessburg et al., 2015).

In 2010, the USDA Forest Service (USFS) initiated the national Collaborative Forest Landscape Restoration (CFLR) program to increase the pace and scale of ecological restoration efforts in western dry conifer and other degraded forests over the next 10 years (www.fs.fed.us/restoration/CFLRP). The program awarded up to $4 million USD annually to selected National Forests (NFs) working

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with collaborative stakeholder groups in landscapes with a pressing need for restoration (Schultz et al., 2012). The expectations and budget of the program included monitoring the effectiveness of the restoration treatments, but for each of the CFLR-funded projects, the NFs and their collaborators were charged with developing their own monitoring programs (Schultz et al., 2014). Few precedents existed to guide the CFLR projects in this endeavor. The monitoring approach utilized by the Fire and Fire Surrogate study (FFS) at 12 sites across the US had some relevance, but FFS treatments tended to emphasize fuels reduction rather than ecological restoration were implemented in 'blocks' with replicated but relatively small treatment and control units (~10 ha) at each site, and were monitored in a relatively standard manner across the sites (Schwilk et al., 2009; Stephens et al., 2012). Numerous other studies have evaluated the effects of restoration treatments on specific ecological properties and processes in western dry conifer forests (e.g., overstory density, Waltz et al., 2003; overstory spatial heterogeneity, Larson and Churchill, 2012; understory plants, Moore et al., 2006; birds, Gaines et al., 2007; mammals, Kalies and Covington, 2012; invertebrates, Waltz and Covington, 2004; and fuel loads, Fulé et al., 2006); however, few have simultaneously evaluated effects on a diverse suite of collaboratively identified properties and processes (Thomas and Waring, 2014).

The Front Range of the southern Rocky Mountains in Colorado was one of the first landscapes in the US to receive CFLR funding in 2010. Large, uncharacteristically severe wildfires occurred in the area in 1996, 2000, and 2002 (Sherriff et al., 2014; Fornwalt et al., 2016), with considerable negative social and ecological impacts (Bhandary and Muller, 2009; Fornwalt et al., 2010; Rhoades et al., 2011; Chambers et al., 2016). These fires catalyzed both a regional fuels reduction effort and the identification of priority landscapes for restoration by a collaborative stakeholder group, the Front Range Roundtable (FRRT; Brown et al., 2001; Culver et al., 2001; FRFTP, 2006). However, relatively few treatments with specifically restoration-oriented objectives were conducted due to limited budgets (Worden and Kleier, 2012; Keely et al., 2013; Ertl, 2015). In 2010, the Arapaho-Roosevelt and Pike-San Isabel NFs worked with the FRRT on a successful application to the CFLR program for funding of restoration treatments across 12,950 ha of NF land designated as highest priority for restoration within the larger 323,750-ha forested landscape (Underhill et al., 2014; Cheng et al., 2015). As the NFs and their collaborators, collectively referred to as the Colorado Front Range Landscape Restoration Initiative (CFRLRI; Underhill et al., 2014), prepared to implement and monitor the CFLR-funded treatments, they expressed numerous questions and concerns regarding the treatments' short- and long-term ecological effects (Schultz et al., 2014; Dickinson et al., 2016).

In coordination with the CFRLRI, we initiated a study in 2011 to evaluate the short-term effects of forest restoration treatments in Colorado's Front Range on several diverse ecosystem components: forest stand- and patch structure, tree regeneration, surface fuels, understory plants, and wildlife use. Several aspects of our study represented important expansions of the monitoring effort planned by the CFRLRI with the budget allocated by the NFs (Clement and Brown, 2011). The original monitoring plan focused on measuring changes in forest stand structure, surface fuels, and tree regeneration in NF treatment units, pre- and post-treatment, via USFS Common Stand Exam (CSE) protocols (Clement and Brown, 2011; USDA Forest Service, 2011). By leveraging supplemental funding, our study was able to extend the scope of the monitoring program in three key ways. First, we included sites on other agencies' lands so that inferences could be extended beyond the NFs. Second, we established pre- and post-treatment monitoring plots not only in treatment units but also in nearby areas not scheduled for treatment, thereby enabling a Before/After/Control/Impact study design (BACI; Stewart-Oaten et al., 1986) that could better evaluate treatment effectiveness (Hutto and Belote, 2013). Third, we supplemented the CSE protocols with new methods to monitor several additional aspects of the ecosystem—understory plants, wildlife use, and forest patch structure—that the CFRLRI found important but had not included in their original monitoring plan due to budget constraints. In this paper, we report on the ecological outcomes of the restoration treatments as well as on the role of our study in both the adaptive management (DeLuca et al., 2010) and adaptive monitoring (Lindenmayer and Likens, 2009) processes. The multiple goals of this project lend it relevance not only to the CFRLRI and the other 22 currently funded CFLR projects, but to many of the additional collaborative restoration and monitoring efforts developing locally, nationally, and internationally in recent years (e.g., Pistorius and Freiberg, 2014 and references therein; JCLRP, 2016).

2. Methods

2.1. Study design

Our study was conducted in three study areas along Colorado's Front Range (Table 1). One study area was located on Boulder County Parks and Open Space land (Heil Ranch; 40°10′N, 105°18′W), one was located on the Roosevelt NF (Estes Valley; 40°16′N, 105°24′W), and one was located on the Pike NF (Phantom Creek; 39°3′N, 105°12′W). Elevations at the study areas averaged approximately 2200 m at Heil Ranch, 2650 m at Estes Valley, and 2830 m at Phantom Creek. Forest overstories at the study areas were dominated or co-dominated by ponderosa pine (*Pinus ponderosa*), with varying proportions of additional overstory species such as Douglas-fir (*Pseudotsuga menziesii*), Rocky Mountain juniper (*Juniperus scopulorum*), and aspen (*Populus tremuloides*) at lower elevation study areas, and lodgepole pine (*Pinus contorta*), subalpine fir (*Abies lasiocarpa*), blue spruce (*Picea pungens*), Engelmann spruce (*Picea engelmannii*), and aspen at higher elevation study areas.

Each of the three study areas contained two to three study sites that consisted of a treatment unit and a nearby paired area not slated for treatment (Table 1). Treatment units ranged in size from approximately 30 to 140 ha. Treatment prescriptions were developed by the respective land management agencies and were necessarily variable due to pre-treatment site conditions, agency regulations, and treatment methods (Fig. 1), but all had the overall goal of meeting ecological restoration objectives and all involved thinning of the forest overstory via specialized machinery or hand crews (Underhill et al., 2014; Nick Stremel, Boulder County Parks and Open Space, Boulder, Colorado, personal communication, 2014). Untreated stands were located within 1 km of treatment units, in comparably sized areas with similar aspect, slope, elevation, soils, and pre-treatment overstory composition and structure.

Plot locations in the treatment units were determined by first visiting coordinates of existing CSE monitoring plots and identifying whether they met the criteria for inclusion in the study (below). If additional plot locations were needed for our study, we subsequently visited coordinates generated using a randomized method in a Geographic Information System (GIS). Because our study focused on evaluating changes in forested stands in ponderosa pine-dominated ecosystems, our criteria for plot selection were that plots contained at least five trees (within a variable-radius plot established using a Basal Area Factor (BAF) of 10, as described below), had at least one ponderosa pine >1.37 m tall, were >250 m from any other established plot and >20 m from a unit edge, and had not been recently disturbed. If these criteria were met at a given location, we established a plot there; if not,

Table 1
Attributes of seven Colorado Front Range study sites at which forest restoration treatments were implemented in ponderosa pine forests in 2011–13. At each site, a diverse suite of metrics was measured in treated and untreated plots before and one year after treatment.

Study site	Treatment description	Treatment date	Treatment area (ha)	Elevation (m)	# of treated plots	# of untreated plots
Heil Ranch (Boulder County Parks and Open Space)						
Heil 5	Mechanical thinning, with slash lopped and scattered	Winter 2012–13	50	1960	6	5
Heil 7	Mechanical thinning, with slash lopped and scattered	Winter 2012–13	70	2100	4	4
Estes Valley (Roosevelt National Forest)						
Estes Valley 13	Hand thinning, with some mastication	Spring 2012	20	2480	3	3
Estes Valley 28	Hand thinning, with slash piled and burned	Winter 2011–12	50	2420	5	4
Estes Valley 34	Hand thinning, with slash piled and burned	Winter 2011–12	10	2160	3	3
Phantom Creek (Pike National Forest)						
Phantom Creek 1	Mechanical thinning, with most slash removed	Summer 2011	60	2740	3	5
Phantom Creek 2	Mechanical thinning, with most slash removed	Summer 2011	150	2630	10	8

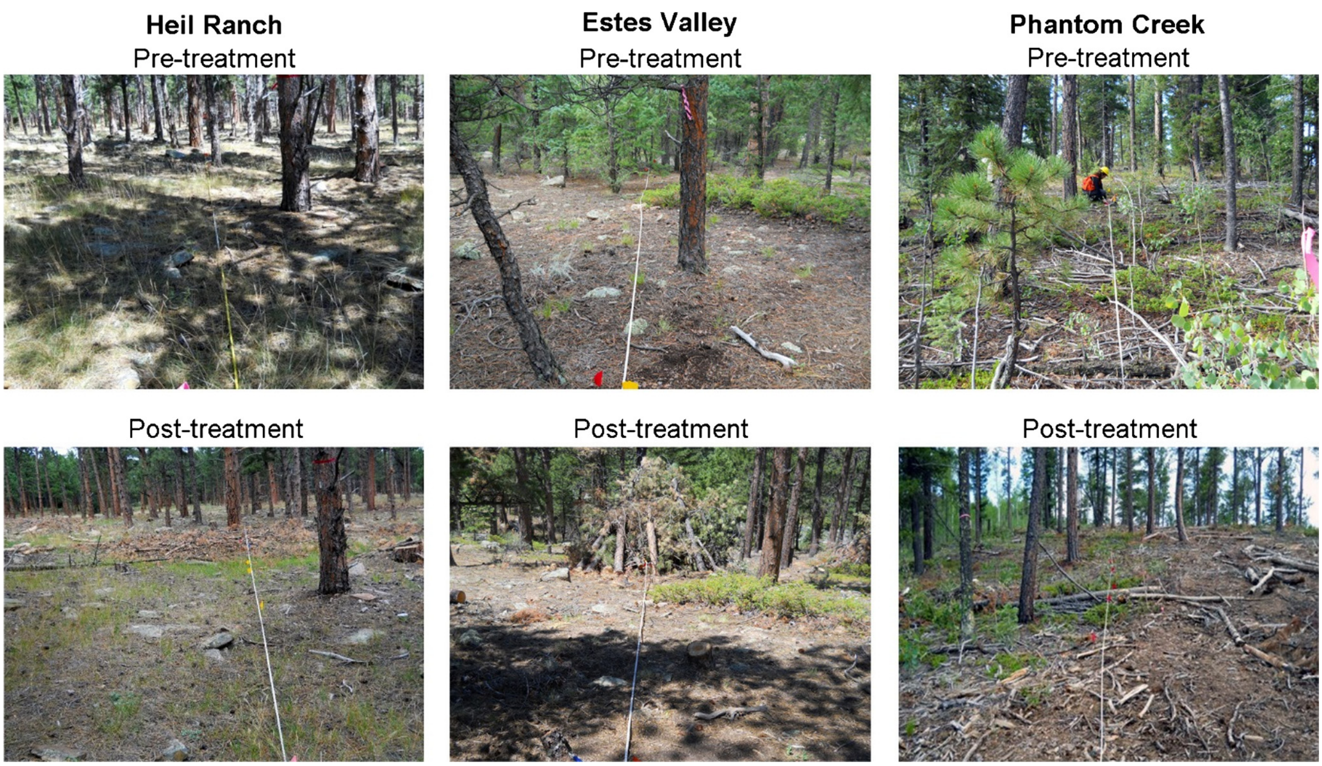

Fig. 1. Representative treated plots before (top) and one year after (bottom) restoration treatments in ponderosa pine forests of the Colorado Front Range.

we moved on to the next set of coordinates until we had established approximately one plot per 5–20 ha within each treatment unit. The number of plots per treatment unit ranged from 3 to 10, and was dependent on the unit's size (Table 1). We established a similar number and density of plots in paired untreated areas as in the treatment units, using the same selection criteria, at coordinates generated randomly in a GIS.

2.2. Sampling methods

We collected pre-treatment data on plots in the treatment units and untreated areas as described below, in summer 2011. All plots were permanently marked to enable resampling at the same locations after treatment. At four of the seven sites, treatments occurred as planned in 2011–12 (Table 1), and we collected post-treatment data there in summer 2012. At the remaining three sites, treatments were postponed by one year, and we collected the post-treatment data in those areas in summer 2013. Our final sample size of plots with both pre-treatment data (2011) and post-treatment data (2012 or 2013) was 66 (34 plots located in a total of seven treatment units, and 32 plots in seven nearby untreated areas).

2.2.1. Forest stand structure

Consistent with the CFRLRI monitoring plan, we established a variable-radius plot (BAF 10) at plot center and measured all live 'in' overstory trees that had a diameter at breast height (dbh; breast height = 1.37 m) of at least 2.54 cm (Clement and Brown, 2011; USDA Forest Service, 2011). For each tree, we recorded species, dbh, and any indications of wildlife use by tree squirrels (e.g., nests, feeding sign at base).

2.2.2. Forest patch structure

To measure forest structure beyond the scale of the plot, we established a sampling transect running 100 m north of each plot center. Along this transect, we recorded via visual inspection the distances covered by closed-canopy forest patches and open patches with no canopy cover directly above the transect line. In this context, we defined closed-canopy forest patches (hereafter referred to as "forest patches") as areas containing sapling (≥1.37 m tall and <2.54 cm dbh) or overstory tree canopies, and we defined open patches as areas with no sapling or overstory tree canopies. If canopies were less than 1.5 m apart, we counted them as part of the same forest patch. Within the forest patches, we noted whether and for what distance the canopy structure was single-storied (i.e., the canopy of only one sapling or overstory tree was present above the transect) or multi-storied (canopies of >1 sapling and/or overstory tree were present above the transect). We truncated the data collected along the 100-m transects prior to analyzing the number and mean lengths of open and forest patches, deleting the measurements of the first and last patches measured on each transect because the locations of the plot centers created arbitrary start and end points. We used the full 100-m data for comparisons of the percent of transect length that was in open-versus forest patches, and the percent of transect length within forest patches that had single-versus multi-storied canopy structure.

2.2.3. Tree regeneration

We counted all regenerating tree seedlings (<1.37 m tall) and saplings present in a 0.002 ha plot (2.5-m radius) around plot center (Clement and Brown, 2011). We identified the species of each regenerating tree and classified its height as <30 cm or ≥30 cm.

2.2.4. Surface fuels

We inventoried surface fuels on one 15.2-m transect per plot, which ran north from plot center. We followed standard protocols (Brown, 1974), and those of the CFRLRI (Clement and Brown, 2011), to tally all downed woody fuels in four size classes: 1-h fuels (<0.64 cm diameter) along a 2.54-m section of the transect; 10-h fuels (0.64–2.54 cm diameter) along a 2.54-m section of the transect; 100-h fuels (2.54–7.62 cm diameter) along a 3.66-m section of the transect; 1000-h fuels (>7.62 cm diameter) along the entire 15.2-m length of the transect. We used allometries to convert the tallies into fuel loads for each of the four fuel size classes, and summed these to yield fine fuel (1-h, 10-h, and 100-h fuels) and coarse fuel (1000-h fuels) loads (Brown, 1974). In addition, we measured the depth of the duff layer and fuel bed (litter plus any wood particles present) at two equally spaced points on the transect.

2.2.5. Understory plants and forest floor substrates

We measured percent cover of vascular understory plants (i.e., forbs, graminoids, shrubs) and forest floor substrates (e.g., litter, soil, wood by size class) for each plot using a point-intercept method. We established four 9.37-m transects in the cardinal directions from plot center. At 100 evenly spaced observation points along each transect, we recorded all forest floor substrates and all understory plants <1.37 m tall. While most plant identifications were made to the species level, some identifications could be made only to the genus level due to difficulties distinguishing species when outside peak morphological development; hereafter, these identifications are also referred to as species. The number of occurrences of each species and substrate was tallied for each transect to calculate percent cover. We also conducted a complete inventory of all understory species present in a 0.04-ha (11.3-m radius) circular plot positioned at plot center. Nomenclature, as well as growth form, nativity, and lifespan classifications, followed the PLANTS Database (NRCS, 2016).

2.2.6. Wildlife use

We searched for and recorded sign from two general "guilds" of wildlife (defined here as groups of species that use resources in a similar way; Simberloff and Dayan, 1991) within a 0.04-ha circular plot located at plot center. These were tree-dwelling squirrels (Abert's squirrel, *Sciurus aberti*, and pine squirrel, *Tamiasciurus hudsonicus*) and ungulates (mule deer, *Odocoileus hemionus*, and elk, *Cervus canadensis*). We tallied sign in numerous categories. For tree squirrels, sign included nests, and feeding sign such as chewed cones, branch clippings, peeled twigs, and middens (Worden and Kleier, 2012); for ungulates, sign included pellet piles, day beds, and browsing on tree trunks. In data analyses, we included only sign that we classified with a high degree of confidence as fresh or active (based on color, texture, and other indications described in training sessions with local wildlife specialists) in the seasons prior to our initial pre-treatment surveys (fall 2010-summer 2011), and in the same seasons following the treatments (fall 2011-summer 2012 or fall 2012-summer 2013, as applicable; Table 1).

2.3. Statistical analyses

We used a Before/After/Control/Impact (BACI) approach to examine the effects of ecological restoration treatments on metrics representing six broad categories: forest stand structure (e.g., total overstory density (trees per ha; TPH)), forest patch structure (e.g., mean length of forest patches), tree regeneration (e.g., density of ponderosa pine regeneration), forest floor substrates and surface fuels (e.g., fine wood loads), understory plants (e.g., total understory plant richness), and wildlife use (e.g., percent of plots with tree squirrel sign).

We used generalized linear mixed models to evaluate the effects of treatment, time, and treatment × time for each metric. Analyses were conducted in SAS 9.4 with the GLIMMIX procedure (SAS Institute Inc., Cary, North Carolina, USA), and used an α of 0.050 to evaluate significance. Models specified the appropriate distribution for each metric (e.g., negative binomial distribution for metrics such as total overstory TPH, lognormal distribution for metrics such as total overstory basal area (BA), beta distribution for metrics such as total understory plant cover). Site and treatment × site were included as random effects. Time was included as a random effect with plot as the repeated measures subject and with the two sampling periods for each plot correlated by a compound symmetry covariance structure. For metrics of tree regeneration, forest floor substrates, surface fuels, understory plants, and wildlife use, measurements in untreated and treated plots occurred both pre-and post-treatment, creating four groups in the data (i.e., untreated plots pre-treatment, untreated plots post-treatment, treated plots pre-treatment, treated plots post-treatment). For these metrics, when treatment × time was significant, we examined pairwise differences between groups using least squares means with a Tukey-Kramer adjustment. For forest stand structure and forest patch structure metrics, there were only three groups in the data (i.e., untreated plots pre-treatment, treated plots pre-treatment, treated plots post-treatment); we did not measure these metrics in untreated plots post-treatment because of limited funding and time and because they were not expected to show measurable change since pre-treatment surveys. Thus, while we could not evaluate the significance of the treatment × time interaction term for these metrics, we still examined pairwise differences between groups using least squares means with a Tukey-Kramer adjustment.

3. Results

3.1. Forest stand structure

The ecological restoration treatments influenced several forest stand structure metrics (Fig. 2). The treatments caused reductions in both total overstory TPH (46% average reduction relative to pre-treatment values) and total overstory BA (36% reduction).

Following treatments, the mean percent of TPH and BA represented by ponderosa pine, however, remained around 77% (p = 0.996 and 0.744, respectively) and the percent represented by Douglas-fir remained around 17% (p = 0.186 and p = 0.242, respectively). The quadratic mean diameter (QMD) of both ponderosa pine and of all species in the overstory increased by around 4 cm (18%) post-treatment.

3.2. Forest patch structure

Several metrics of forest patch structure were also altered by the treatments (Figs. 3 and 4). The percent of transect length composed of forest patches in treatment units decreased by an average of 45% following treatments, and there was a corresponding, significant increase in the percent of transect length composed of open patches. However, the mean number of both open- and forest patches remained relatively constant in treatment units, with between 5 and 6 open- and forest patches per transect, both pre- and post-treatment. The mean length of both the open- and forest patches changed significantly in treated stands. Mean forest patch length decreased by 54%, to 5.5 m, and mean open patch length increased by 62%, to 14.3 m. Within forest patches, the mean percent of transect length composed of single-storied and multi-storied canopy was almost equal prior to treatment, but after treatment, the mean percent of single-storied canopy increased by over a third, to 83% of forest patch length.

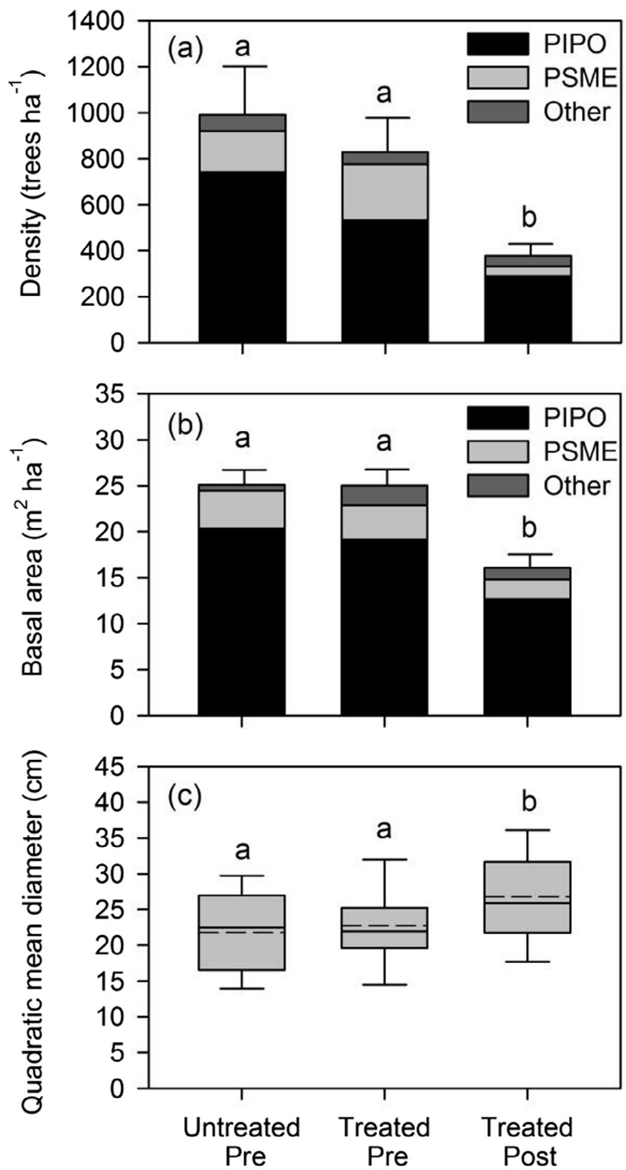

Fig. 2. Mean overstory (a) density (stems (trees) per ha), (b) basal area, and (c) quadratic mean diameter in ponderosa pine forests of the Colorado Front Range that experienced restoration treatments in 2011–13. Data were collected at 34 treated plots and 32 untreated plots in seven treatment units and seven nearby untreated areas. Untreated plots were measured pre-treatment and treated plots were measured pre- and post-treatment. Values for density and basal area are further differentiated for ponderosa pine (PIPO), Douglas-fir (PSME) and other species (e.g., aspen, blue spruce, subalpine fir); standard errors are for all species combined. Box-and-whisker diagrams for quadratic mean diameter show the mean (dashed line), median (solid middle line), 25th and 75th percentile (solid bottom and top line), and 10th and 90th percentile (bottom and top whiskers) values. For each metric, groups that share letters were not significantly different ($\alpha = 0.050$).

3.3. Tree regeneration

Metrics of tree regeneration showed no changes on treated or untreated plots in the first year post-treatment (Table 2). Across all plots and years, 58% contained seedlings or saplings of any tree species, and the mean density of this regeneration was 3939 stems ha^{-1}. Meanwhile, 41% of plots contained ponderosa pine regeneration, with a mean of 2200 stems ha^{-1} found across all plots and years. The mean density of the smallest size class of conifers (<30 cm tall) was 3386 stems ha^{-1} across all plots and years; ponderosa pine represented the majority (54–78%) of these small seedlings.

3.4. Forest floor substrates and surface fuels

The cover of litter/duff, soil, and fine wood on the forest floor was altered by the treatments, but no other metrics of substrate or wood cover, load, or depth were affected (Table 3). Prior to treatment, litter and duff covered 85% on average of each plot's surface, and following treatments this value dropped by 10% on treated plots only. Cover of fine wood nearly doubled on treated plots after treatment, from an average of 10% to 18%, but the loading of fine wood did not likewise change. Soil cover also nearly doubled on treated plots after treatment, from an average of 3% to 5%.

3.5. Understory plants

Restoration treatments had little impact on understory plant communities in the first post-treatment year (Table 4). Of the 16 understory plant metrics examined here, only one—shrub cover—exhibited a relatively clear treatment effect, with cover on treated plots decreasing from 7% pre-treatment to 4% post-treatment. Across all plots and years, plots contained a total of 31 understory species with a total cover of 11%. No single growth form dominated both total richness and cover; total richness was dominated by forbs (63% of total), while total cover was dominated by shrubs (52% of total). Understory plant communities were, however, highly native-dominated, with native species comprising 94% of total richness and 95% of total cover. Furthermore, species with long life spans were considerably more abundant (88% and 96% of total richness and cover, respectively) than those with short life spans. The five most commonly encountered species in our plots were the native long-lived forbs pineywoods geranium (*Geranium caespitosum*), goldenrod species (*Solidago* spp.), and pussytoes species (*Antennaria* spp.), and the native long-lived graminoids prairie Junegrass (*Koeleria macrantha*) and sedge species (*Carex* spp.).

3.6. Wildlife use

The percentage of plots with signs of recent tree squirrel use decreased one year after treatment, but this decline occurred

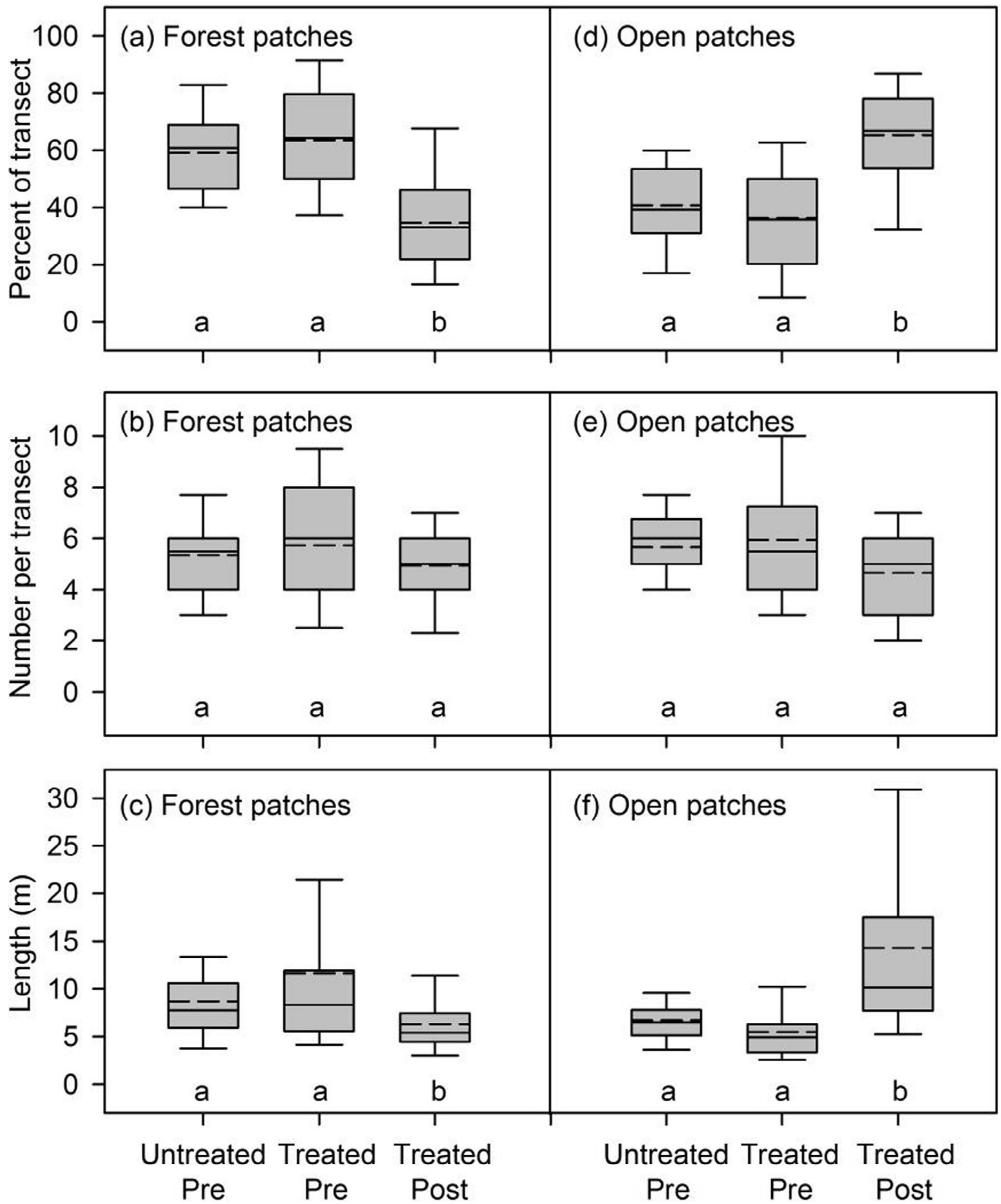

Fig. 3. Attributes of (a, b, c) forest patches and (d, e, f) openings (i.e., unforested patches) along 100-m transects in ponderosa pine forests of the Colorado Front Range that experienced restoration treatments in 2011–13. Transects in treatment units (n = 33) were measured pre- and post-treatment, and transects in untreated areas (n = 32) were measured pre-treatment. Box-and-whisker diagrams show the mean (dashed line), median (solid middle line), 25th and 75th percentile (solid bottom and top line), and 10th and 90th percentile (bottom and top whiskers) values. For each metric, groups that share letters were not significantly different (α = 0.050).

across both treated and untreated plots (on average, 46% fewer plots had tree squirrel sign in 2012/13) and thus reflected interannual change rather than change associated with treatments (Table 5). The abundance of tree squirrel sign was quite variable (up to a maximum of 490 freshly harvested cone cobs and needle clippings on one plot) and was also lower across all plots in 2012/13. In contrast, fresh ungulate sign was consistently present on around 25% of all plots, both pre- and post-treatment.

4. Discussion

The ecological restoration treatments in these Colorado Front Range ponderosa pine forests caused marked short-term (i.e., one year post-treatment) changes to several metrics representing forest stand- and patch structure. However, we found little change in other ecological components—few significant increases or decreases were detected in the numerous metrics of tree regeneration, surface fuel loading, understory plant communities, and wildlife use that we examined. Three components of forest floor cover experienced changes, but the degree of change may not have been ecologically significant. To varying extents, these results were in line with the desired effects of the treatments for the CFRLRI, and with the findings of other studies. Below we discuss each suite of metrics in turn. We also discuss how this study has contributed to the CFRLRI's adaptive management process, and explore the overall relevance of collaborative adaptive monitoring to future forest restoration efforts.

4.1. Forest stand structure

We found that treatments caused a mean decrease in total overstory TPH of almost 50%, a mean decrease in total BA of almost 40%,

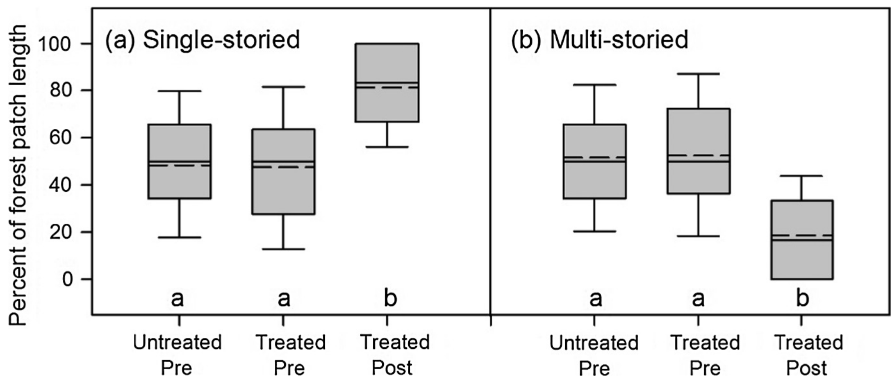

Fig. 4. The percent of the forest patches (see Fig. 3) along 100-m transects that had (a) single-storied and (b) multi-storied canopy structure in ponderosa pine forests of the Colorado Front Range that experienced restoration treatments in 2011–13. Transects in treatment units (n = 33) were measured pre- and post-treatment, and transects in untreated areas (n = 32) were measured pre-treatment. Box-and-whisker diagrams show the mean (dashed line), median (solid middle line), 25th and 75th percentile (solid bottom and top line), and 10th and 90th percentile (bottom and top whiskers) values. For each metric, groups that share letters were not significantly different (α = 0.050).

Table 2

Means (and standard errors) of tree regeneration metrics before and one year after restoration treatments in ponderosa pine forests of the Colorado Front Range; n = 34 treated plots and n = 32 untreated plots in seven treatment units and seven nearby untreated areas. No interactions of treatment × time were significant for any metric at α = 0.050, so pairwise comparisons between groups were not performed.

Metric	Untreated		Treated		Treatment	Time	Treatment × time
	Pre	Post	Pre	Post		P-value	
Density (stems ha^{-1})							
All regeneration	5579 (1940)	4911 (2076)	2823 (868)	2442 (731)	0.542	0.651	0.977
Ponderosa pine	3559 (1399)	2595 (987)	1404 (601)	1243 (567)	0.648	0.576	0.516
Douglas-fir	1740 (1040)	2113 (1595)	790 (339)	629 (362)	0.250	0.922	0.480
Conifers < 30 cm tall	5175 (1914)	4553 (2051)	2018 (750)	1799 (690)	0.547	0.728	0.980

Table 3

Means (and standard errors) of forest floor substrate and surface fuel metrics before and one year after restoration treatments in ponderosa pine forests of the Colorado Front Range; n = 34 treated plots and n = 32 untreated plots in seven treatment units and seven nearby untreated areas. Significant (α = 0.050) p-values are shown in bold. For metrics where treatment × time was significant, pairwise comparisons between groups were evaluated using least squares means; values sharing letters were not statistically different.

Metric	Untreated		Treated		Treatment	Time	Treatment × time
	Pre	Post	Pre	Post		P-value	
Forest floor substrate cover (%)							
Litter/duff	85.3 (1.5)[ab]	82.5 (1.5)[ab]	85.0 (1.5)[a]	75.3 (1.7)[b]	0.256	**<0.001**	**0.012**
Fine wood	11.9 (2.0)[ac]	7.3 (0.9)[b]	10.4 (1.0)[ab]	18.3 (2.0)[c]	0.083	0.589	**<0.001**
Coarse wood	1.6 (0.3)[a]	1.3 (0.3)[a]	1.5 (0.2)[a]	2.3 (0.4)[a]	0.294	0.492	**0.026**
Soil	3.0 (0.9)[a]	3.0 (0.9)[a]	3.0 (0.7)[a]	4.9 (0.8)[b]	0.760	0.050	**0.044**
Surface fuel load (Mg ha^{-1})							
Fine wood	1.4 (0.2)	1.0 (0.2)	1.6 (0.3)	1.3 (0.2)	0.476	0.299	0.833
Coarse wood	1.8 (0.6)	1.5 (0.5)	2.4 (0.6)	2.2 (0.5)	0.271	0.750	0.596
Surface fuel depth (cm)							
Duff	1.2 (0.2)	2.1 (0.3)	1.0 (0.1)	2.1 (0.3)	0.936	0.281	0.890
Fuel bed	4.6 (0.5)	3.3 (0.4)	4.0 (0.4)	3.2 (0.5)	0.820	**0.004**	0.666

and a mean increase of QMD of almost 20%. These changes met the objective of the CFRLRI to reduce forest density, and are within the range of changes made by restoration treatment projects in other ponderosa pine forests (e.g., Fulé et al., 2001, 2006; Waltz et al., 2003; Youngblood et al., 2006; Schwilk et al., 2009; Roccaforte et al., 2010, 2015). Some of these other projects aimed to shift forest conditions toward specific target levels, commonly based on reference conditions representing forest density documented in the area prior to European-American settlement (e.g., Roccaforte et al., 2010). Although the CFRLRI is quantifying pre-settlement forest density and other overstory metrics in the Front Range to inform further discussion and planning of restoration objectives (e.g., Dickinson, 2014; Brown et al., 2015), it has not adopted explicit targets at this time. One aspect of treatments did not appear to represent the CFRLRI's current objectives for overstory conditions: the percentage of total TPH and BA comprised of ponderosa pine remained constant post-treatment, on average, rather than increasing. Around 60% of treatment plots had mixed species composition, and of those, approximately half experienced an increase in their percentage of ponderosa pine BA post-treatment (mean 14.2 ± 3.5%) while half experienced a similar decrease in ponderosa BA (mean 14.3 ± 3.8%). Approximately 25% of treatment plots contained 100% ponderosa pine both before and after they were thinned, and the remaining 15% of plots had 100% ponderosa pine but experienced no change in BA during treatment. In future work, we suggest that both restoration actions and monitoring methods consider starting conditions—and, if possible, the age and canopy status of removed and residual trees—in more detail than was

Table 4
Means (and standard errors) of understory plant metrics before and one year after restoration treatments in ponderosa pine forests of the Colorado Front Range; n = 34 treated plots and n = 32 untreated plots in seven treatment units and seven nearby untreated areas. Significant (α = 0.050) p-values are shown in bold. For metrics where treatment × time was significant, pairwise comparisons between groups were evaluated using least squares means; values sharing letters were not statistically different.

Metric	Untreated		Treated		Treatment	Time	Treatment × time
	Pre	Post	Pre	Post		P-value	
Richness (species 0.04 ha^{-1})							
Total plant	32.4 (1.3)	31.4 (1.5)	29.3 (1.1)	30.3 (1.5)	0.753	0.840	0.333
Forb	21.0 (0.8)^{a}	19.3 (0.9)^{a}	18.1 (0.9)^{a}	19.0 (1.0)^{a}	0.154	0.470	**0.020**
Graminoid	5.9 (0.5)	6.5 (0.6)	5.8 (0.5)	6.3 (0.6)	0.759	0.075	0.738
Shrub	4.9 (0.3)	4.9 (0.4)	4.4 (0.4)	4.3 (0.4)	0.159	0.546	0.546
Exotic plant	1.3 (0.3)	1.4 (0.4)	1.4 (0.4)	2.0 (0.6)	0.907	0.412	0.305
Native plant	30.9 (1.1)	29.9 (1.2)	27.6 (1.0)	28.1 (1.1)	0.077	0.694	0.189
Short-lived plant	3.2 (0.6)^{a}	2.5 (0.6)^{b}	2.4 (0.4)^{ab}	3.6 (0.8)^{ab}	0.752	0.243	**0.007**
Long-lived plant	28.6 (1.1)	28.1 (1.2)	25.9 (0.9)	26.0 (0.9)	0.106	0.721	0.587
Cover (%)							
Total plant	13.3 (1.8)	13.0 (1.7)	11.1 (1.3)	8.7 (1.2)	0.091	0.088	0.166
Forb	2.2 (0.6)	2.2 (0.5)	1.4 (0.2)	1.3 (0.3)	0.079	0.792	0.863
Graminoid	4.0 (0.7)	4.1 (0.7)	3.0 (0.5)	3.3 (0.7)	0.195	0.654	0.846
Shrub	7.1 (1.4)^{a}	6.7 (1.3)^{ab}	6.5 (1.1)^{a}	3.9 (0.7)^{b}	0.173	**<0.001**	**<0.001**
Exotic plant	0.6 (0.4)	0.6 (0.3)	0.3 (0.2)	0.3 (0.1)	0.671	0.601	0.651
Native plant	12.6 (1.8)	12.3 (1.6)	10.7 (1.3)	8.3 (1.1)	0.096	0.074	0.135
Short-lived plant	0.6 (0.3)	0.6 (0.3)	0.1 (0.0)	0.3 (0.2)	0.901	0.153	0.232
Long-lived plant	12.7 (1.8)	12.4 (1.7)	10.9 (1.3)	8.2 (1.1)	0.105	0.050	0.110

Table 5
Means (and standard errors) of wildlife use metrics before and one year after restoration treatments in ponderosa pine forests of the Colorado Front Range; n = 34 treated plots and n = 32 untreated plots in seven treatment units and seven nearby untreated areas. Values represent fresh or active signs of use recorded for 2 guilds of species (tree squirrels and ungulates). Significant (α = 0.050) p-values are shown in bold. No interactions of treatment × time were significant for any metric, so pairwise comparisons between groups were not performed.

Metric	Untreated		Treated		Treatment	Time	Treatment × time
	Pre	Post	Pre	Post		P-value	
Plots with sign (%)							
Tree squirrels	93.8 (4.3)	65.6 (8.5)	70.6 (7.9)	26.5 (7.7)	0.060	**<0.001**	0.828
Ungulates	25.0 (7.8)	21.9 (7.4)	23.5 (7.4)	29.4 (7.9)	0.625	0.862	0.524
Number of signs (0.04 ha^{-1})							
Tree squirrels	66.1 (12.7)	27.8 (8.3)	29.7 (12.1)	23.4 (14.7)	0.159	**0.010**	0.612

possible within the scope of our study, to permit more focus on the composition, structure, and possible successional trajectories of stands post-treatment.

4.2. Forest patch structure

Following treatments, we found significant increases in the degree of overall openness of stands, and an increase in the mean and variability of length of open patches, but no change in the mean numbers of open- or forest patches. These findings indicate that treatments generally focused on extending the length of existing open patches, causing a corresponding decrease in the length of forest patches—rather than, for example, adding new open patches (which would have led to an increase in the mean number of open patches) or combining existing open patches (which would have led to a decrease in the mean number of open patches). Following treatment, the mean length of forest patches became highly consistent (6.3 ± 0.5 m) as did the lengths of open patches (mean 14.3 ± 2.0 m), indicating that the treatments created somewhat homogenous stand structure. Implementation therefore likely continued some elements of the regular-spacing prescriptions that were characteristic of traditional fuels reduction treatments on NFs in the Front Range, rather than meeting the CFRLRI's objective to increase heterogeneity in stand structure (Underhill et al., 2014; Dickinson et al., 2015). The collection and interpretation of data, both historical and current, on spatial configuration of trees and of forest- and open patches openings within stands is a relatively new area of work within the forest restoration community (e.g., Sánchez Meador et al., 2011; Larson and Churchill, 2012; Churchill et al., 2013; Tuten et al., 2015; Dickinson et al., 2016) but is motivated by extensive evidence that spatial patterns of forest canopy at diverse scales affect processes such as fire, hydrologic cycles, seed dispersal, understory development, and wildlife habitat use (Larson and Churchill, 2012 and references therein.) Our novel transect sampling method allowed us to quantify some aspects of forest spatial stand structure beyond the scale of plots in a more rapid and cost-effective manner than the detailed stem- or stand-mapping conducted as part of traditional spatial pattern analyses; this method may prove useful in other monitoring efforts (see Davis et al., 2016; Dickinson et al., 2016).

Treated stands had significantly less multi-storied stand structure (∼15%) than untreated or pre-treatment stands (∼50%), raising some concerns within the CFRLRI. Although multi-storied forest patches have often been viewed as containing undesired 'ladder fuels' that traditional fuels reduction treatments aimed to remove, more recent ecological restoration prescriptions for uneven-aged management suggest retaining more multi-storied groups of trees, interspersed with openings of variable sizes that would not be likely to carry high-severity crown fire across uncharacteristically large areas (Larson and Churchill, 2012; Churchill et al., 2013). Some multi-storied patches might experience torching if an ignition occurred, but if enough treeless openings and low-density stands were present, this structure would be expected to promote desired outcomes such as a mosaic of fire behavior and effects (Churchill et al., 2013); resilience to disturbances such as insect epidemics that often target certain size classes or species of trees (Fettig and McKelvey, 2014); and retention of diverse habitat in the multi-storied patches for certain species of wildlife (e.g., Germaine et al., 2004; Fontaine and Kennedy, 2012). In conjunction with field trips and examination of aerial imagery, our data helped CFRLRI silviculturists adapt some aspects of treatment prescriptions to increase the heterogeneity of post-treatment forest structure instead of simplifying it (Jeff Underhill, USFS, Denver, Colorado, personal communication, 2016).

4.3. Tree regeneration

There were no significant effects of treatment on tree regeneration, which was not wholly unexpected in the first post-treatment year. Ponderosa pine has episodic regeneration patterns throughout its range (Cooper, 1960; Bailey and Covington, 2002; Shepperd et al., 2006) and there were no years during the time period of our study with the combinations of high cone production and adequate precipitation patterns that facilitate establishment of seedlings (Shepperd et al., 2006; Flathers et al., 2016). Other studies of ponderosa regeneration patterns after forest thinning in Colorado (Shepperd et al., 2006; Ertl, 2015), as well as in Arizona (Bailey and Covington, 2002; Puhlick et al., 2012), Montana (Fajardo et al., 2007), and New Mexico (Thomas and Waring, 2014), have found significantly greater densities in thinned stands versus unthinned stands. However, these studies occurred later post-treatment than our study, providing more time for trees to respond to changes in growing conditions. Although the CFRLRI has not yet identified objectives for tree regeneration, the current density of ponderosa pine regeneration that we measured on treated plots was more than twice the minimum of 470 seedlings ha^{-1} recommended for 'stocking' in ponderosa forests of the Front Range, and the percentage of treated plots with ponderosa pine regeneration that we found—35%—may increase over time to the level of 70% that is specified in current forest management plans as a post-treatment objective after three to five years (PSICC, 1984; ARP, 1997).

4.4. Forest floor substrates and surface fuels

Litter is very abundant in Colorado Front Range ponderosa pine forests, and although its cover on plots decreased significantly after treatments, its depth did not change and the decrease in cover from 85% to 75% may not be ecologically significant. Soil exposure experienced a similarly modest increase, from 3% to 5%, which may limit opportunities for the post-treatment establishment of understory plants and ponderosa pine seedlings (Cooper, 1960; Xiong and Nilsson, 1999; Bonnet et al., 2005). Desired conditions for cover of litter and soil have not yet been addressed by the CFRLRI. Percent cover of another forest floor layer, fine wood, nearly doubled as a result of the treatments, consistent with the results of a recent meta-analysis that found increases in surface fuels after thinning treatments in many dry conifer forest types (Fulé et al., 2012). However, we did not also find statistically significant changes in fine fuel loads, suggesting that either fine wood biomass did not increase along with wood cover, or that our measurement protocols for fine fuel loads were not sensitive enough to detect some potentially ecologically significant changes. In fact, we noted considerable amounts of new downed wood in 31 of the 34 treated plots (91%) after treatment, in the form of tree boles, slash, and/or chips (Fig. 1). However, these new pieces of wood fell on our single fuels transect in only 10 of 34 plots (29%). We suggest using more extensive sampling and/or more thorough methods to document changes in fuel loading in future (e.g., Keane and Gray, 2013), given the CFRLRI's objective that treatments should reduce, or at least not increase, the abundance of fine or coarse downed wood in stands (Clement and Brown, 2011).

4.5. Understory plants

Understory plant metrics of richness and cover were largely unimpacted by restoration treatments in the first post-treatment year. This finding is in agreement with similar short-term (1–2 years post-treatment) studies that have been conducted in dry conifer forests of the Colorado Front Range (Ertl, 2015) and elsewhere in the US (Abella and Springer, 2015 and references therein). To some extent, this lack of change may be considered in line with CFRLRI project objectives (Clement and Brown, 2011); for example, treatments did not promote an undesirable increase in the (already low) abundance of exotic plants, and did not negatively affect understory plant diversity. Furthermore, longer-term (5+ years post-treatment) studies commonly document an increase in many understory plant metrics as understory plants respond to the reduction in competition with overstory trees, suggesting that we may see positive understory plant responses in our plots in the future (Laughlin et al., 2006; Thomas and Waring, 2014; Abella and Springer, 2015 and references therein; Ertl, 2015; Fornwalt et al., 2017). Longer-term post-treatment increases in metrics such as native species richness and cover would be viewed as furthering CFRLRI project objectives, while increases in other metrics, particularly exotic species richness and cover, would not be. It is critical to continue monitoring these plots so that the longer-term effectiveness of the treatments can be evaluated.

4.6. Wildlife use

For tree squirrels, metrics of use changed significantly with time but not with treatment; for ungulates, the percentage of plots with recent sign remained consistent across treatment and time. These results support those of other studies in similar forest types, both those that found annual changes in occupancy by tree squirrels (e.g., Wampler et al., 2008; Worden and Kleier, 2012) and those that found consistent use of treated areas by ungulates (Germaine et al., 2004; Thomas and Waring, 2014). However, given the large spatial and long temporal scales at which wildlife species respond to changes in habitat structure, food supply, and abiotic factors (Kalies et al., 2010), it is likely premature to evaluate whether treatments have met the CFRLRI's general objective that treatments promote, or at least do not reduce, the occurrence of native wildlife species expected to use restored forest habitat (Clement and Brown, 2011). Our findings demonstrated to concerned stakeholders in the CFRLRI that tree squirrels and ungulates did not avoid treated areas of the forest, but our monitoring methods did not permit a full evaluation of these species' abundance, behavior, or distribution. Recently, more detailed studies of Abert's squirrel, for example, have identified variations in post-treatment habitat use associated with many factors (season, year, and distinct attributes of trees or overstory; Dodd et al., 2006; Loberger et al., 2011; Worden and Kleier, 2012; Yarborough et al., 2015), suggesting that focused monitoring of this ponderosa pine-obligate species over longer time frames may be warranted to evaluate any changes in its use of forests treated by the CFRLRI.

4.7. This study's contribution to the adaptive management process of the CFRLRI

Taken together, our results suggest that these restoration treatments in the Colorado Front Range achieved, in the first year, progress toward several of the desired conditions and trends identified by the CFRLRI as management goals (Clement and Brown, 2011; Underhill et al., 2014; Dickinson et al., 2015). However, as described above, some of our results suggested the need for modification of restoration treatment prescriptions in an active adaptive management framework (DeLuca et al., 2010; Aplet et al., 2014). For certain metrics, such as forest stand structure, prescriptions have indeed been adapted over time to foster greater variability in treated stands, although this is an ongoing process (Underhill et al., 2014; Dickinson et al., 2015) that has also benefited from recent stand reconstruction surveys that identified more open and heterogeneous historical forest conditions than those we measured post-treatment (Dickinson, 2014; Brown et al., 2015). In general, our study represented successful effectiveness monitoring, and helped catalyze important discussions by the CFRLRI of shared objectives for restoration, monitoring, decision-making, and thresholds for action in the adaptive management process (Aplet et al., 2014).

A primary contribution of our study to adaptive management was in the 'adaptive monitoring' sub-process of the cycle (Lindenmayer and Likens, 2009, 2010; Aplet et al., 2014). By catalyzing at least three important expansions and revisions of the CFRLRI's monitoring protocol, this project exemplified several hallmarks of successful adaptive monitoring. First, our adoption of a BACI study design that included untreated areas highlighted the fact that some metrics—such as understory plants, wildlife habitat use, and tree regeneration—can change annually, which may preclude a valid assessment of treatment effects from comparisons made only between pre- and post-treatment data in treated areas (Block et al., 2001; Larson et al., 2013). Our BACI framework and results set an important precedent for comparative monitoring of changes in untreated areas, which the CFRLRI has continued. Second, we identified metrics and methods that have been adopted as longer-term components of an expanded monitoring program funded by the CFRLRI. Our data on understory plants, in particular, spurred the CFRLRI to develop additional understory plant-related questions and plans, and our general methodology was applied across a much larger spatial scale beginning in 2015. Finally, the methods that we suspected were not adequate for a valid assessment of metrics of interest have been revised either moderately (in the case of surface fuels) or significantly (a greatly expanded program of wildlife monitoring began in 2014). If our study had not developed additional monitoring efforts complementary to the initial efforts of the CFRLRI, it is possible that several important metrics might not have been addressed or measured until much later in the lifecycle of the CFLR program.

4.8. The role of landscape-scale collaborative restoration projects in fostering resilient forests

Although short-term, this study's inclusion of a diverse suite of ecological metrics in its evaluation of CFLR treatments represents a valuable contribution to collaborative efforts to incorporate science into restoration-focused management of dry conifer forests of the western US (Allen et al., 2002; Larson et al., 2013). Monitoring to evaluate both the intended ecological effects of treatments, and any unanticipated negative consequences (Hutto and Belote, 2013), will be critical as innovative, broad-scale restoration efforts like the CFLR have increasing impacts across forest landscapes (DeLuca et al., 2010; Schultz et al., 2014; Davis et al., 2016; JCLRP, 2016). Our study has drawn on two decades of relevant work conducted in dry conifer forests of other western regions (e.g., Covington et al., 1997; Allen et al., 2002; Stephens et al., 2012; Hessburg et al., 2015) and has also identified some new opportunities for collaborative monitoring efforts (Davis et al., 2016). The challenges we encountered in integrating diverse perspectives on monitoring priorities were worthwhile steps in the process of building an expanded long-term monitoring program. Tracking a suite of ecosystem components that not only represent the initial outcome of the treatment (e.g., total overstory TPH, length of forest patches, fuel loads) but also reflect spatially and/or temporally dynamic processes (e.g., habitat use by wildlife, development of understory plant communities, forest patch structure associated with past or future fire behavior, tree regeneration) will allow a more holistic understanding of forests' ecological resilience to future disturbances such as wildfire and changing climate. Equally important, monitoring of large-scale forest restoration work by collaborative groups facilitates a focus on socially and ecologically important components of the forest that will contribute to more effective shared stewardship of forest resources and ecosystem services over time.

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