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Effects of collaborative monitoring and adaptive management on restoration outcomes in dry conifer forests

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ABSTRACT

In response to large, severe wildfires across the western US, federal initiatives have been enacted to increase the pace, scale, and quality of ecological restoration in fire dependent forests. To address uncertainty and controversy in agreement among specific restoration prescriptions on national forest land, several initiatives adopt a collaborative adaptive management (CAM) strategy wherein monitoring data can inform stakeholder input into future management actions. It is unclear, however, how such approaches may change restoration outcomes. Here we assess the extent to which CAM strategies impact restoration outcomes that were implemented as part of the Collaborative Forest Landscape Restoration Program (CFLRP) in ponderosa pine-dominated forests of the Colorado Front Range. We assessed stand-level desired conditions across 24 projects over a 7-year period to determine how restoration treatments contribute to desired conditions, and compared treatment outcomes over implementation time to assess whether the CAM processes contribute towards treatments better approximating restoration. We found that restoration treatments improve aspects of forest structure related to stand density. However, meeting objectives related to forest composition and horizontal structural complexity goals were not met. Additionally, CAM processes were effective at improving outcomes related to forest density over implementation time, but novel tools and approaches may be required so that outcomes related to forest composition and horizontal structural complexity are more congruent with restoration objectives. Evaluating the success and challenges of CAM provides insight to improve collaborative and large-scale restoration.

1. Introduction

Climate change, anthropogenic stressors, and shifting disturbance regimes are altering forest structure, composition, and functioning globally (Anderson-Teixeira et al., 2013; Prichard et al., 2017; Seidl et al., 2017; McDowell et al., 2020). High severity wildfires that cause widespread tree mortality, coupled with climate change are altering dry-conifer forests in the western US, even causing conversion from forests to grasslands and shrublands (Stevens-Rumann et al., 2018; Davis et al., 2019; Coop et al., 2020). Over 150 years of logging, grazing, land use development, and forest fire suppression have disrupted fire regimes characterized by frequent, low-intensity surface fires (Covington et al., 1994; Allen et al., 2002; Peterson et al., 2005; Parker et al., 2006; Baker et al., 2007; Hicke et al., 2015). The lack of widespread use of prescribed fire and landscape level treatments as a management tool, coupled with fire suppression has contributed to increases in forest density, fuel loads, and fire-intolerant tree species; thereby, increasing fire intensity and severity when fires occur under extreme weather conditions (Stephens et al., 2013; Calkin et al., 2015; Hessburg et al., 2019).

To address this situation, many scientists and managers implement thinning and prescribed fire to restore the structure and composition associated with historically frequent fire regimes in seasonally-dry forests, and lower the vulnerability to high tree mortality from fire (Covington, 2000; Brown et al., 2004; Kaufmann et al., 2005; Stephens et al., 2020). Translating these general silvicultural principles into specific plans and prescriptions for forest restoration has been beset by uncertainty and controversy regarding the appropriate type, size, intensity, and potential impacts of proposed activities (DeLuca et al., 2010). Examples of concerns voiced both locally and throughout western fire dependent forests regarding the use of mechanical tree harvesting to

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achieve restoration objectives on national forest lands managed by the US Forest Service (USFS) include applying older fuels reduction methods to newer ecological restoration projects, using clearcutting as a restoration method, the retention of old trees, and applying similar fine-scale spatial heterogeneity principles repeatedly at the landscape scale (Cosgrove, 2003; Coughlan, 2003; Vaughn and Cortner, 2005; Burns and Cheng, 2007).

To address controversy and uncertainty in the management of fire-prone forests, individuals and organizations have turned to collaborative processes, wherein stakeholders work through differences, engage in social learning about issues and concerns, and identify mutually desirable objectives that are achieved through collective action (Vosick, 2016; Schultz and Butler, 2019). Collaborative adaptive management (CAM) is one such process where management activities are conceived as learning opportunities from which to gather and analyze data and observations on management effects, interpret results, and evaluate the need for and type of subsequent management adjustments (Allen and Garmestani, 2015). Essential to adaptive management success is a monitoring strategy identifying restoration goals, metrics, and methods of evaluation (Lyons et al., 2008; Cundill and Fabricius, 2009; DeLuca et al., 2010; Davis et al., 2015).

CAM to mitigate wildfire risks in dry forests of the western US has been facilitated by notable policy changes such as the Collaborative Forest Landscape Restoration Program (CFLRP). The US Congress established the program in 2009 to focus federal investments towards priority National Forest System lands in need of forest restoration through a competitive allocation process (Omnibus Public Lands Act, 2009; Schultz et al., 2012). The program requires multi-party monitoring and an adaptive management strategy to gauge the projects progress towards collaboratively-defined restoration goals and desired conditions throughout the life of the program (Schultz et al., 2014). In this regard, the CFLRP presents a unique opportunity to examine the extent to which forest restoration treatments changed over implementation time as a result of CAM.

Here, we assess the efficacy of these informal processes in influencing adaptive management of forest restoration treatments over implementation time. We present pre-treatment and 1–2 year post treatment results from effectiveness monitoring measurements collected for 24 projects over a 7-year timeframe associated with the Colorado Front Range CFLRP (FR-CFLRP). Previous studies examined the effects of forest restoration treatments on forest structure (Underhill et al., 2014; Dickinson et al., 2016; Cannon et al., 2018), fire behavior (Ziegler et al., 2017), understory plant communities (Briggs et al., 2017), and avian and small mammal species (Latif et al., 2020) in the same landscape. However, these studies present results from a fixed timeframe of treatments (e.g., treatment effects from a single year of treatment) or do not explicitly account for changes in treatment design. In this paper, we build on previous studies to examine how restoration outcomes change over the lifespan of a large-scale restoration program by tracking changes in restoration outcomes implemented over a 7-year period. Understanding how treatments change over implementation time can shed light on the effectiveness of a programmatic CAM approach for forest restoration. Specifically, we assess outcomes related to stand-level forest structure and species composition over implementation time as they relate to FR-CFLRP's monitoring plan (Clement and Brown, 2011; Barrett et al., 2018). When restoration goals were developed for the FR-CFLRP's monitoring plan, specific targets were not defined (e.g. residual target basal area range), rather, trends were developed to describe successful projects. Guidelines for restoration outlined in the monitoring plan include further increasing the abundance of ponderosa pine (*Pinus ponderosa* Douglas ex P. Lawson & C. Lawson) relative to other conifer species, particularly on wet aspects where Douglas-fir (*Pseudotsuga menziesii* Mirb.) has traditionally not been removed as much as monitoring results have suggested; removal of small diameter ladder fuels; and enhancing the heterogeneity of forest density and size throughout stands (Clement and Brown, 2011; Dickinson et al., 2014; Barrett et al., 2018; Cannon et al., 2018). However, as projects developed throughout the lifetime of the project, more general guidelines have been established such as target residual basal areas by forest type ranging from 9 to 14 m^2 ha^{-1} in ponderosa pine forest types to 18 to 23 m^2 ha^{-1} in more moist mixed conifer stands (Underhill et al., 2014). Although several reports and studies evaluate early treatment outcomes of the FR-CFLRP (Young et al., 2013; Dickinson et al., 2016; Briggs et al., 2017; Cannon et al., 2018), to date, no program-scale evaluation of the evolution of treatment outcomes over implementation time has been undertaken. Given the FR-CFLRP's adaptive management framework, we hypothesize that treatment outcomes will converge on desired conditions over implementation time, with more recent treatments more closely reflecting desired conditions compared to earlier treatments.

2. Methods

2.1. Study area

The montane forests of the Colorado Front Range are complex and interact strongly with aspect and elevation. Ponderosa pine is dominant in the lower montane zone (roughly 1800 m to 2400 m, Kaufmann et al. 2006), with increasing dominance of Douglas-fir at higher elevations and more mesic aspects and topographic features. These dry mixed-conifer forests may also contain aspen (*Populus tremuloides* Michx.), limber pine (*Pinus flexilis* James) and lodgepole pine (*Pinus contorta* Douglas ex Loudon), though typically in lower abundances (Peet, 1981; Addington et al., 2018). Upper montane and wet mixed-conifer forests generally begin around 2400 m, with increased abundances of lodgepole pine and Engelmann spruce (*Picea engelmanii* Parry ex Engelm), and lower abundances of ponderosa pine. Soils along the Front Range are variable, but are generally immature and shallow, with parent materials consisting of granite, gneiss, and schist (Johnson and Cline, 1965; Addington et al., 2018). Climatic conditions are also variable across the front range, with an annual precipitation of 25–50 cm (Addington et al., 2018), and temperatures generally decreasing from south to north (Barry, 1992; Addington et al., 2018). Restoration activities under the FR-CFLRP commonly take place within or near the wildland urban interface on the Arapaho and Roosevelt National Forests and Pawnee National Grassland (ARP), and the Pike and San Isabel National Forests and Cimarron and Comanche National Grasslands (PSICC).

2.2. Data compilation

To assess treatment efficacy and outcomes over implementation time, we compiled available monitoring data from the USFS on forest structure and composition for all sites both before and after treatments. Data was collected using standard USFS Common Stand Exam (CSE) protocols (USDA Forest Service, 2019) within 1–2 years before and after USFS treatments, providing a robust dataset of pre- and post-treatment structural and compositional metrics. Plot surveys included inventory of overstory trees (with diameter at breast height [DBH] > 12.7 cm) using variable radius plots and smaller trees up to 1.37 m tall in three 16.2 m^2 subplots. For each tree, species and DBH was recorded. We excluded snags and regenerating trees (DBH < 4 cm) from analyses to make results comparable with related studies (Battaglia et al., 2018; Cannon et al., 2018). Exact treatment delineations sometimes changed following monitoring, thus we included only plots that fell within harvested treatment unit boundaries. Additionally, we removed treatment units that had less than five plots from our analysis. Our final data encompassed 24 projects, representing 72 treatment units, and 776 plots ranging from 2010 to 2017 (Fig. 1, Table 1), each with pre- and post-treatment data, totaling 1552 observations.

We summarized plot data within each treatment unit to obtain estimates of the following metrics: (1) stem density, (2) basal area, (3) quadratic mean diameter (QMD), and (4) the importance value of ponderosa pine and Douglas-fir (Curtis and McIntosh, 1951).

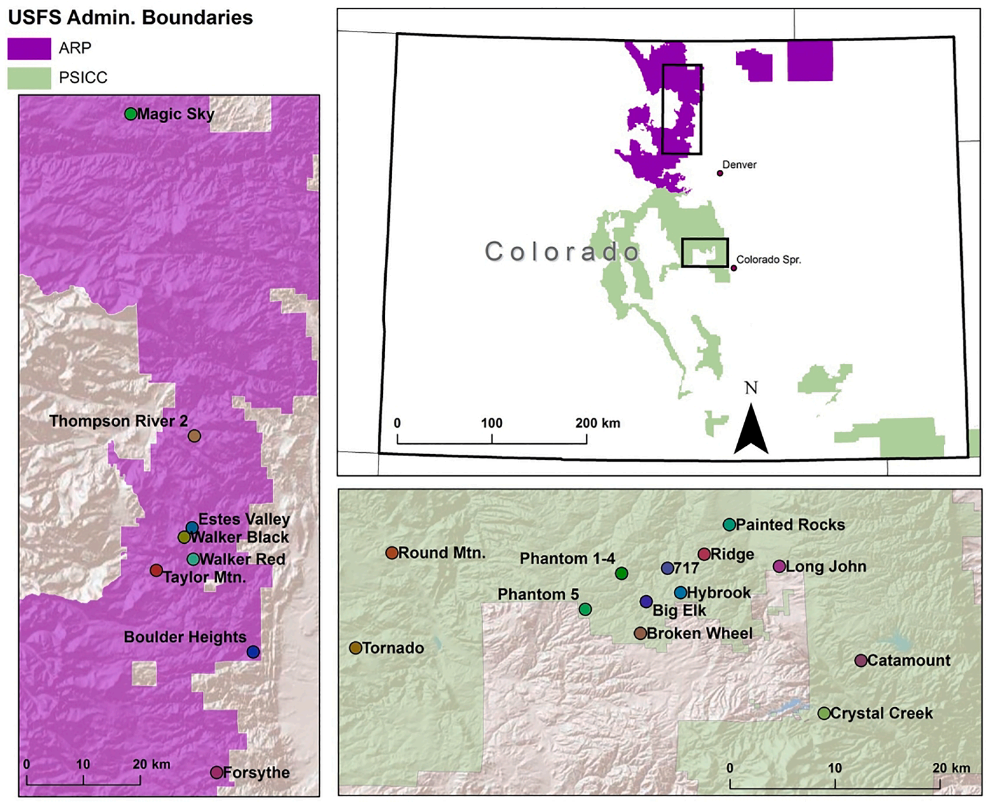

Fig 1. Map of projects included in this study, including 8 projects from the Arapaho and Roosevelt National Forests and Pawnee National Grassland (ARP), and 16 from the Pike and San Isabel National Forests and Cimarron and Comanche National Grasslands (PSICC), Colorado, USA.

Table 1
Forest characteristics of monitoring areas in Arapaho and Roosevelt National Forests and Pawnee National Grassland (ARP) and Pike and San Isabel National Forests and Cimarron and Comanche National Grasslands (PSICC) in the Colorado Front Range, including project names, years, treatment size, number of treatment units, and number of plots per project analyzed. Accomplished year refers to the year that contracting of the proposed work was complete, reflecting the degree to which treatments were affected by adaptive management.

Forest	Project	Accomplished Year	Avg. Elevation (m)	Treatment Area (ha)	# Units Monitored	# Plots Monitored
PSICC	Phantom 1	2010	2695	91	1	7
ARP	Taylor Mountain	2010, 2011	2680	164	1	12
ARP	Walker Red	2010, 2011	2558	276	7	131
ARP	Estes Valley	2011	2380	311	5	69
PSICC	Phantom 2	2011	2695	340	3	61
PSICC	Phantom 3	2011	2695	266	1	36
ARP	Walker Black	2011	2344	54	1	7
ARP	Boulder Heights	2012	1897	46	1	6
PSICC	Catamount	2012	2648	127	3	31
PSICC	Long John	2012, 2013	2616	120	3	27
PSICC	Phantom 4	2012	2695	143	3	25
PSICC	Big Elk	2013, 2014	2677	79	2	17
PSICC	Broken Wheel	2013	2758	65	2	13
PSICC	Crystal Creek	2013	2548	166	8	46
ARP	Forsythe	2014	2400	190	3	37
PSICC	Ridge	2014, 2015	2506	243	9	76
PSICC	717	2015	2547	271	3	41
ARP	Magic Sky	2015	2154	46	1	10
PSICC	Hybrook	2016	2576	180	7	56
PSICC	Painted Rocks	2016	2483	122	1	5
PSICC	Phantom 5	2016	2829	105	1	12
ARP	Thompson River 2	2011, 2016	2512	280	1	6
PSICC	Tornado	2016	2758	77	2	17
PSICC	Round Mountain	2017	2614	81	3	28

Importance value is calculated as the sum of relative frequency, relative density, and relative abundance and ranges from 0 (least dominant), to 300 (most dominant). Ponderosa pine importance value was calculated for treatment units that had ponderosa pine present prior to treatment (n = 72 treatment units) for pre- vs. post-treatment analyses.

To assess changes in horizontal complexity we used Shannon's equitability index (E_H). This metric is often used to assess evenness in species assemblages but can be applied to other aspects of forest structure to assess complexity (Turner et al., 2001; Valbuena et al., 2012). We binned basal area into six classes of equal width (0–11.5, 11.5–23.0, 23.0–34.5, 34.5–46.0, 46.0–57.5, and > 57.5 m^2 ha^{-1}). We then calculated E_H using proportions of the stand in each basal area class using the following equation:

$$E_H = \frac{-\sum_{i=1}^{n} p_i ln p_i}{ln n}$$

where p_i is the proportion of each stand in basal area class i and n is the number of classes, resulting in an estimate of within-stand horizontal complexity ranging from 0 to 1 with 1 representing even representation among the six density classes (Shannon and Weaver, 1963; Pielou, 1966). Values closer to 1 better reflect desired conditions for horizontal complexity, as restoration outcomes would have created a more even distribution of diameter classes rather than narrowing in on one residual target basal area. To help visualize changes in horizontal complexity over implementation time, we created basal area distributions for treatment units representing early (2010–2013) and late (2014–2017) projects. We also calculated solar radiation (Fu and Rich, 2002) using ArcGIS 10.4.1 for each plot to assess varying treatment outcomes on drier versus wetter sites. This measurement accounts for atmospheric conditions, elevation, aspect, and shading from nearby topography, and we use it as a proxy for soil moisture which is a key driver in vegetation communities on the Front Range (Peet, 1981). We considered all sites above the median solar radiation of 1612 kW-h m^2 as drier and those below the median as wetter sites following Cannon et al. (2018).

We were unable to obtain individual prescriptions and marking guidelines for each project analyzed as part of this project. However, projects generally aimed to increase spatial heterogeneity from the sub-stand (<0.4 ha) to landscape level (>4047 ha) by creating heterogeneous stand structures containing individual trees, clumps of trees, and interspersed openings of various densities across the landscape (Underhill et al., 2014). Projects have generally transitioned toward operational select methods of harvesting, in which the contractor selects trees based to cut based on quantitative and qualitative target residual metrics, to the combined use of tree marking in more complex treatments while still implementing operator select methods in less complex prescriptions (Underhill et al., 2014; Dickinson and Cadry, 2017).

2.3. Data analysis

To assess the effectiveness of restoration treatments in meeting collaborative goals, we summarized plot level data into treatment units (n = 72) for analysis. We used paired t-tests to test for pre- versus post-treatment differences in tree density, basal area, QMD, importance value, and horizontal complexity index using R (R Core Team 2019). To assess how CAM may lead to changes in restoration outcomes (residual tree density, basal area, QMD, and horizontal complexity) over implementation time, we used a linear mixed effects model with implementation year as a fixed effect, and project as a random effect to account for the fact that treatments within the same project may have similar outcomes. We used the lme4 (Bates et al., 2015) and lmerTest (Kuznetsova et al., 2017) in R to create mixed effects models. To help visualize trends in restoration outcomes over time, we used linear regression to build relationships between each treatment outcome and implementation time, however all results were interpreted from linear mixed effects models. While it could be informative to compare treatments conducted without CAM with treatments conducted with CAM, all treatments analyzed in this project were implemented with collaborative monitoring. Therefore, we assume that more recent projects were more heavily influenced by CAM due to the cumulative knowledge learned by the collaborative over the lifetime of the FR-CFLRP's monitoring program. To determine if CAM had an effect on ponderosa pine or Douglas-fir importance value, we used linear regression on the difference of importance value for a given species (pre-treatment vs post-treatment) using year accomplished as the predictor variable. Year accomplished is reported in the USFS Forest Activities and Tracking System database and refers to year that contracting of proposed work was complete. This date may be earlier than the date in which treatments were actually implemented; however, because treatment prescriptions are relatively inflexible after contracting, this date may better reflect the degree to which treatments were influenced by adaptive management. We calculated percent reduction in basal area and forest density in addition to absolute differences to account for sites that may have been relatively low density prior to restoration treatments. We removed aspen from analysis, and only report densities of conifer species because aspen is not commonly targeted for removal for restoration treatments in the region.

To address if restoration treatments favored ponderosa pine averaged across all projects, particularly on wetter aspects as is stated in collaborative restoration goals, we used a linear mixed effects model with importance value as the response variable, aspect (wet vs. dry) and treatment (pre- vs post-treatment), and their interaction as predictor variables, and a random site effect. We used the lme4 (Bates et al., 2015) and lmerTest (Kuznetsova et al., 2017) in R, and made post-hoc comparisons with the emmeans (Lenth, 2019) package. When aspect was a significant predictor of importance value, we used additional linear mixed effects models with treatment unit response as the response variable, a fixed effect for accomplishment year, and a random effect for project to test if the change in ponderosa pine importance value or Douglas-fir importance value resulting from treatment changed over implementation time on sites that were considered more mesic (n = 36 treatment units in mesic sites). We used linear regression in Fig. 4 to help visualize main effects, however all statistical inferences were made from linear mixed effects models.

3. Results

Averaged across all projects, restoration treatments accomplished many of the objectives with respect to overall overstory reduction and resulted in a 42% reduction in basal area (from 21.4 m^2 ha^{-1} to 12.0 m^2 ha^{-1}; t = 14.26, $p < 0.01$, Fig. 2A) and a 63% reduction in tree density (from 649 ha^{-1} to 243 ha^{-1}; t = 13.19, $p < 0.01$, Fig. 2B). Restoration increased QMD from 23.5 cm to 28.2 cm (t = -11.85, $p < 0.01$, Fig. 2C). However, treatments did not affect the importance value of ponderosa pine (t = − 1.37, p = 0.18, Fig. 2D). Additionally, equitability of basal area classes among treatment units decreased 17%, from 0.53 to 0.38 (t = 8.02, $p < 0.01$). Pre- and post-treatment basal area distributions for early (2010 – 2013) and late (2014 – 2017) projects show truncated distributions post-treatment relative to pre-treatment across treatment units. In both early and late projects, post-treatment basal area distributions were truncated around 25 $m^2 ha^{-1}$, and were more heavily distributed at lower basal areas (Fig. 3), explaining the decrease in equitability of basal area classes resulting from treatment.

Both residual basal area (p = 0.01) and tree density ($p < 0.01$) decreased with accomplishment year. Additionally, percentage of basal area removed (p = 0.04, Fig. 4A) increased with accomplishment year, and density reduction had a marginally significant trend with accomplishment year (p = 0.06, Fig. 4B) which resulted in larger QMD's in later treatments ($p < 0.01$; Fig. 4C). Horizontal complexity of treatment units decreased more in recent treatments relative to older ones ($p < 0.01$; Fig. 4D). Additionally, adaptive management did not have an effect on increasing ponderosa pine importance value from older versus more recent treatments (p = 0.48). The importance value of ponderosa pine was higher for drier sites compared to wetter sites ($p < 0.01$), with an estimated difference of 51.9 ± 8.9 (SE). However, we did not find a significant aspect × treatment interaction (p = 0.86), indicating that treatments did not change importance values differently on drier and wetter sites. Additionally, we did not detect that importance value changed over implementation time for either ponderosa pine (p = 0.75, Fig. 4E) or Douglas-fir (p = 0.65, Fig. 4F) on wetter sites.

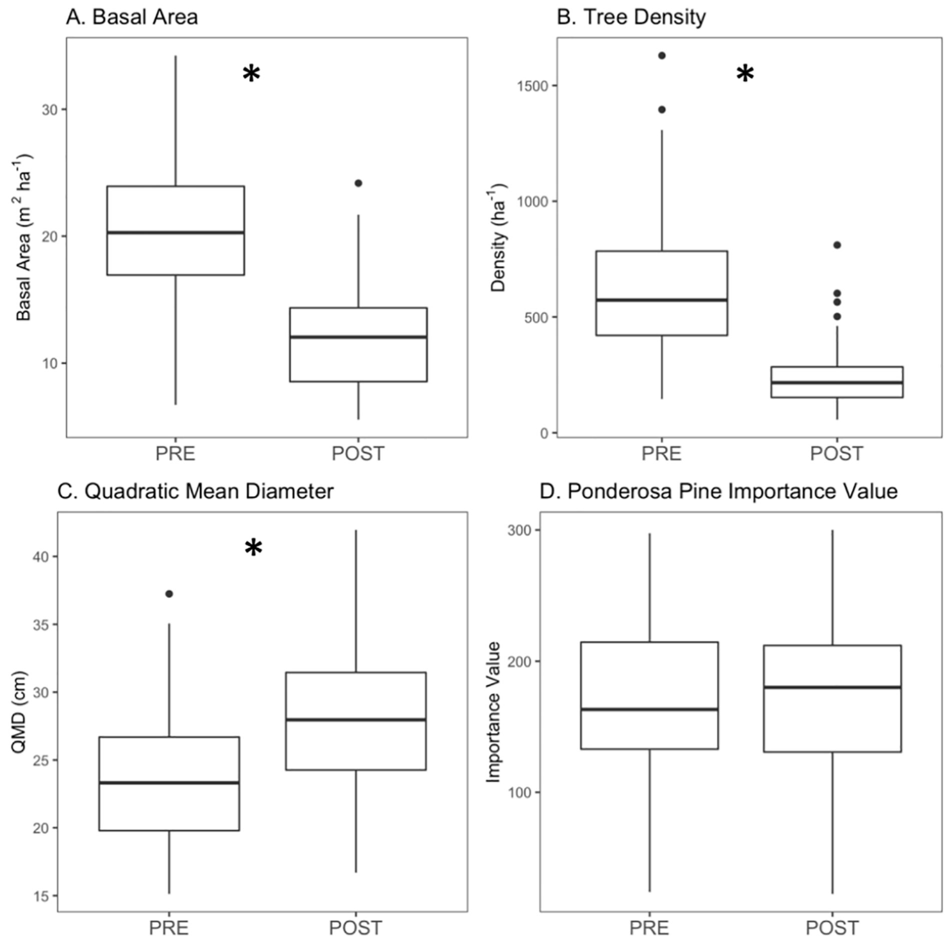

Fig 2. Pre- and post-treatment changes in (A) basal area, (B) tree density, (C) QMD and (D) ponderosa pine importance value. Asterisks denote significant differences using paired t-tests and $\alpha = 0.05$. For all significant comparisons (A-C), $p < 0.01$, and $p = 0.18$ for ponderosa pine importance value (D).

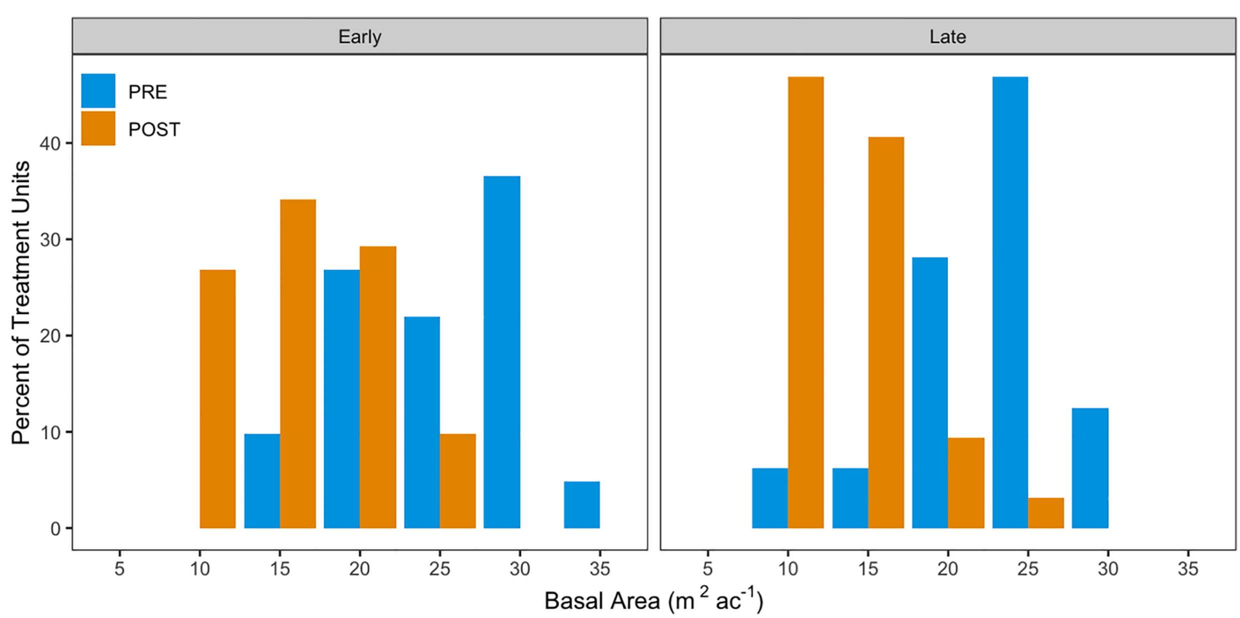

Fig 3. Pre- and post-treatment basal area distributions for treatment units for early (2010–2013, left panel) and late (2014–2017, right panel) projects.

4. Discussion

Results from this study are consistent with other studies in the western U.S., in which density targets are generally met, but goals related to species composition (Ziegler et al., 2017; Lindsay and Johnston, 2019) and spatial heterogeneity (Churchill et al., 2013; LeFevre

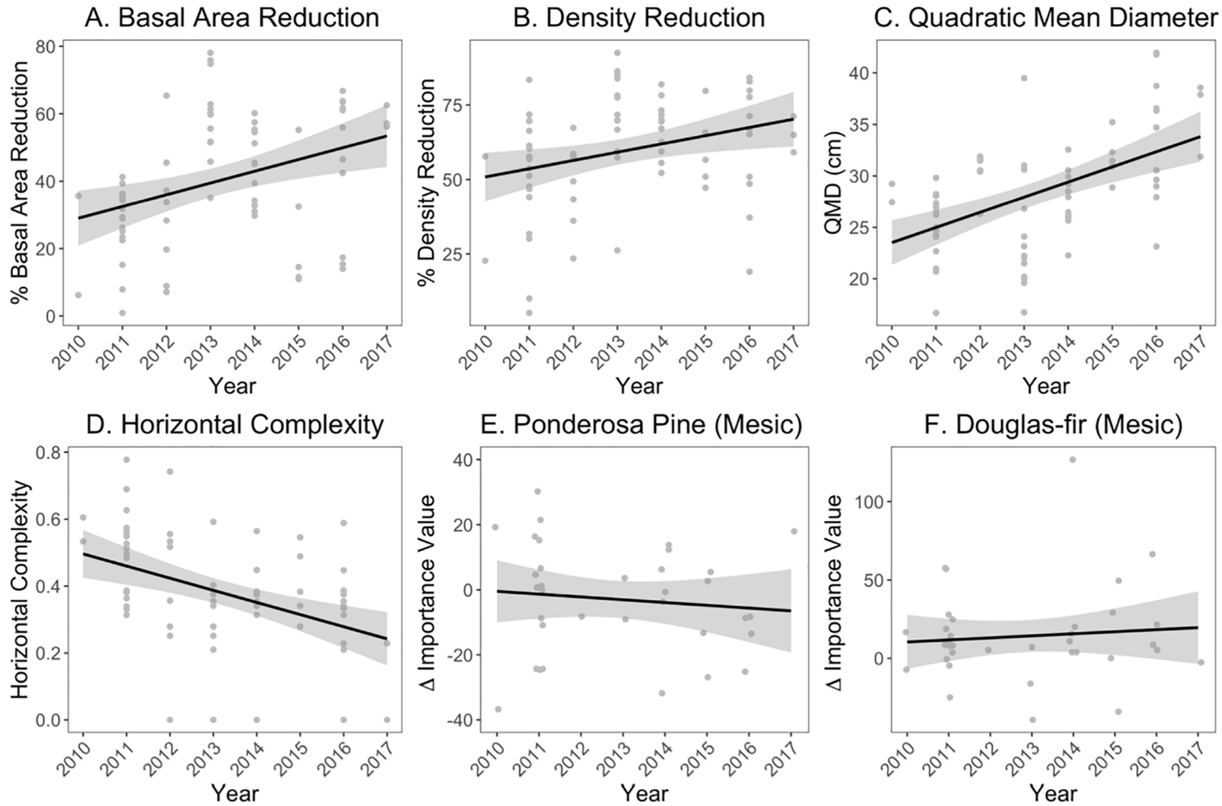

Fig. 4. Linear regression showing the effect of treatment accomplishment year on (A) percent basal area reduction, (B) percent tree density reduction, (C) quadratic mean diameter, (D) horizontal complexity, (E) changes ponderosa pine importance value on wetter sites, and (F) changes in Douglas-fir importance value on wetter sites. The shaded gray area represents a 95% confidence interval. Overall results included project-level effects from linear mixed effects models, but are not shown here.

et al., 2020; Lindsay and Johnston, 2020; Maher et al., 2019) are often not met. Similar to our findings, Ziegler et al. (2017) found that despite a preference for retaining ponderosa pine, restoration treatments did not dramatically change species composition in the southern Rocky Mountains, and Lindsay and Johnston (2020) did not detect shifts in species composition from shade tolerant to more fire tolerant species in the eastern Oregon. Additionally, Lindsay and Johnston (2020) found that treatments generally resulted in even-spaced tree structure even though variability was intentionally built into prescriptions. Similarly, LeFevre et al. (2020) found that despite restoration treatments achieving density, and in this case, composition targets, residual stands were either more uniform or more clumped compared to historical conditions in northeast Washington. Achieving spatially heterogeneous tree structure is a common challenge throughout the west with unique approaches to address the challenges including tailored prescriptions to incorporate historical spatial patterns in tree marking guidelines (Churchill et al., 2013) and the use of tablets to monitor progress towards prescription targets in real time (Maher et al., 2019). Our study builds on this knowledge by exploring changes in treatments over implementation time and suggests that CAM may lead to large changes in density related outcomes while other outcomes related to spatial pattern and composition may be recalcitrant. Overall, we found that restoration treatments changed forest structure toward desired conditions for some, but not all collaboratively defined objectives. Consistent with early results reported from the FR-CFLRP, restoration activities are making progress towards some non-spatial goals focused on reducing tree densities and increasing QMD's (Briggs et al. 2017; Cannon et al. 2018). Throughout the lifetime of the project, The FR- CFLRP adaptive management process indicated further reductions in forest densities may be required to achieve restoration objectives (Cannon and Barrett, 2016; Addington et al., 2018). The inclusion of an additional 18 projects and 4 years of CSE data relative to Cannon et al. (2018) indicates an additional 10% reduction in tree density and 7% reduction in stand basal areas were achieved. This, coupled with the trends in the proportion of tree density and basal area removed over implementation time (Fig. 4A and 4B) suggests that feedback from the adaptive management process may have had an impact on forest structure. Additionally, Cannon et al. (2018) reported no effect of treatment on QMD, while we found an effect of restoration on residual QMD, and higher residual QMD in more recent treatments (Fig. 4C), indicating CAM may have had an effect of adaptive management on QMD.

Although restoration contributed to desired changes in non-spatial structure, outcomes related to changes in forest composition diverged from goals as outlined by the CAM process. While a greater diversity of tree species should be retained on mesic sites (Dickinson et al., 2014), a greater reduction of Douglas-fir was desired given historical reconstructions (Battaglia et al., 2018; Cannon et al., 2018). We did not detect an effect of restoration treatments on the proportion of ponderosa pine in a stand across all of the CFLRP treatments. This indicates that restoration treatments were not effective at increasing the relative abundance of ponderosa pine regardless of the year a project was implemented across both wetter and drier sites. Additionally, we did not detect an effect of adaptive management on the proportion of ponderosa pine or Douglas-fir on wetter sites, where we expected the proportion of ponderosa pine to increase at the expense of Douglas-fir, especially on wetter sites and at later implementation times given the feedback generated from the CAM process. These results are also similar with those reported by Briggs et al. (2017), in which the percentage of ponderosa pine and Douglas-fir remained relatively constant pre- and post-treatment.

Historically, forest restoration of fire-prone forests across the west have not met goals involving spatial heterogeneity of forest structure at both stand and landscape scales (Churchill et al., 2013; Gillson et al., 2019; LeFevre et al., 2020; Maher et al., 2019). Restoration treatments on the Front Range did not increase within-stand horizontal complexity in project areas as expected based on desired conditions (Dickinson et al., 2014). Instead, we found a 21% decrease in horizontal complexity as a result of restoration, suggesting a narrower range of basal area classes resulting from management. Additionally, we found a negative relationship between horizontal complexity and accomplishment year, suggesting that more recent treatments resulted in lower horizontal complexity relative to older treatments. This result may indicate that as treatments removed greater basal area over time the resulting treatment units were more narrowly distributed around a similar target basal area. This suggests that in restoration treatments aiming to decrease tree density, there is a risk of also reducing within-stand heterogeneity if spatial patterns are not explicitly considered. Lindsay and Johnston (2019) made a similar observation in which tree markers and operators may be selecting trees to create similar residual target densities at fine spatial scales, thereby reducing residual heterogeneity of tree structure at coarser spatial scales. Many challenges surround achievement of spatial outcomes of restoration treatments, and they can vary based on the prescription and implementation method used (Underhill et al., 2014; Dickinson and Cadry, 2017; Cannon et al., 2018). Unfortunately, we were unable to obtain individual prescriptions and implementation methods for projects included in the study, which could have provided a more nuanced analysis of treatment outcomes. However, implementation methods have trended to include more individual tree marking for more complex prescriptions while still using operator select methods on less complex prescriptions (Underhill et al., 2014; Dickinson and Cadry, 2017). Methods to explicitly maintain heterogeneity, such as pre-planning gaps and skips based on productivity gradients (Addington et al., 2018), explicit tracking of groups and openings during the tree marking process (Maher et al., 2019) or use of landscape planning tools designed to maintain heterogeneity (Cannon et al., 2020) may also be critical to achieving more nuanced objectives so that goals can be achieved at both the stand and landscape scales. With regards to the restoration objectives that were not met, it is worth noting that this analysis only considered the first entry of sites, and subsequent entries and the potential use of prescribed fire could further move stands towards desired conditions. Additionally, it should be noted that CSE data was only analyzed for areas that were actively managed, and ignores non-managed stands where basal areas are higher. Consideration of larger landscapes which contain treated and untreated stands may lead to better insight of how treatments are affecting landscape complexity at larger scales.

5. Management implications

Overall, restoration treatments implemented by the FR-CFLRP were successful at reducing forest densities, and CAM contributed to treatments approaching desired conditions with regard to forest density over time. However, achieving goals related to horizontal complexity and the associated adaptive management processes remains a challenge. Discussions with foresters that implemented many of the analyzed projects revealed several constraints that may have led to undesirable treatment outcomes. Early in the project, many foresters felt conflicts between restoration goals for projects and the guidelines approved in previous National Environmental Policy Act (NEPA) documents for the area that emphasized fuel reduction objectives rather than collaboratively discussed restoration objectives (Cannon et al., 2018). It is common for implementers to feel constrained from institutional barriers or legal requirements such as NEPA documents that have not kept pace with evolving ecological understanding of the systems to which they apply (Thrower, 2006; Benson and Stone, 2013; Scarlett, 2013; Beratan, 2014). These voiced constraints about the NEPA process are congruent with some of the challenges discussed in other studies (e.g. Benson and Garmestani, 2011; Benson and Stone, 2013; Schultz et al., 2012) in which NEPA documents need to be flexible and site-specific enough to meet restoration objectives for specific projects while embracing new understandings of social and ecological systems throughout the lifetime of the program. Individual tree marking was commonly cited as the most desired method of prescription implementation to achieve complex objectives; however, cost constraints typically result in Operator Select methods in which the operator selects trees based on a description of the trees to remove or a description of the desired outcomes of a prescription including residual forest structure. Operator Select methods inherently lead to more interpretation by operators regarding which trees to cut, which may influence project outcomes in terms of residual stand composition and the distribution of residual basal areas across project areas. This is consistent with results discussed in Dickinson and Cadry (2017), in which foresters and operators both preferred individual tree marking because it removes uncertainty of which trees should be cut. An alternative solution may be a hybrid approach using individual tree marking to mark specific features such as openings while Operator Select may be used between features (Dickinson and Cadry, 2017), or real-time implementation monitoring with tablet applications to guide marking crews in prescription implementation (Maher et al. 2019). Overall, we feel that CAM can help improve outcomes related to density reduction, but other outcomes related to species composition and structural complexity may require novel tools and approaches to achieve objectives.

CRediT authorship contribution statement

Kevin J. Barrett: Conceptualization, Methodology, Software, Formal analysis, Investigation, Data curation, Writing - original draft, Visualization. **Jeffery B. Cannon:** Conceptualization, Validation, Investigation, Writing - review & editing, Supervision. **Alex M. Schuetter:** Conceptualization. **Antony S. Cheng:** Writing - review & editing, Supervision, Project administration, Funding acquisition.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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