Integrating Fuels Treatments and Ecological Values in Piñon-Juniper Woodlands: Fuels, Vegetation, and Avifauna

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Jonathan D. Coop, Center for Environment and Sustainability, Western State Colorado University, Gunnison CO 81231, jcoop@western.edu

Patrick A. Magee, Natural and Environmental Sciences, Western State Colorado University, Gunnison CO 81231, pmagee@western.edu

Abstract

Mastication and hand-thinning treatments are increasingly utilized by land managers as a means of reducing tree cover for fire hazard mitigation and other habitat objectives in piñon-juniper (P-J) woodlands. However, the effects of these treatments on ecological processes including fire, and on a wide range of species, particularly vulnerable P-J obligate birds, are incompletely understood. To address these knowledge gaps we measured vegetation and fuels, and conducted bird point counts at 232 sites in 29 pairs of 1-11-year-old treatments and untreated adjacent controls in P-J woodlands of the Arkansas River valley in central Colorado. We used a suite of statistical approaches including paired t-tests, mixed-effects models, and occupancy analysis, to assess treatment impacts on vegetation, fuels, and bird occupancy. We also developed fire behavior models to examine expected treatment impacts on fire behavior along gradients of fuel loads and under varying fuel moisture scenarios.

Treatments drove major, persistent ecological shifts relative to controls. Tree cover and canopy fuels were reduced; concomitantly, down woody surface fuels, forb, and grass cover increased. Treatments exhibited rapid, large, and persistent increases in the frequency, richness, and cover of 20 non-native plant species including cheatgrass (*Bromus tectorum*). Exotic plant expansion appears linked to the disturbance associated with treatment activities, reductions in tree canopy, and alterations to ground cover. Effective mitigation of non-native plant species may necessitate additional pre- and post-treatment control measures. Treatments substantially reduced the occupancy of piñon-juniper specialist and conifer obligate bird species including the Virginia's Warbler and Gray Flycatcher at the landscape scale and the Piñon Jay at the local scale. Bird species that used open and edge habitats such as the Mountain Bluebird and the Lark Sparrow increased in occupancy within treatments.

Decreased canopy fuels and increased herbaceous surface fuels including exotic annuals are expected to alter potential fire behavior via reduced active crown fire probability, but also increased surface fire intensity, flame length, and rate of spread. Modeled fire behavior suggests that under most conditions, treatments will be highly effective at reducing active crown fire risk, but also that treatments generally removed more trees than necessary to mitigate this risk. Retention of more trees within treatments will benefit P-J obligate birds. Fire behavior models also indicated that residual trees were still highly susceptible to passive crown fire (torching), particularly in masticated sites. Models including simulated treatment modifications suggest that this remaining crown fire risk could mostly be eliminated via follow-up surface fuel reductions (e.g., Rx fire) and/or crown base height increases (e.g., pruning). As such, we propose that managers consider additional post-treatment fuels interventions to increase the fire resistance of residual trees, which is anticipated to yield both social and ecological benefit. We encourage managers carrying out P-J mastication projects to explicitly consider 1) potential trade-offs between desired treatment outcomes and potentially unwelcome impacts, and how these might be mitigated, and also 2) whether or not tree removal treatments may be warranted given anticipated climate change impacts to these woodlands.

Background

Land management actions inevitably entail trade-offs between suites of social and ecological values. In the western United States, increasing wildfire activity and suppression costs are increasingly impelling the broad-scale implementation of a range of fuel reductions treatments intended to decrease wildfire hazard. Where historical land use and fire suppression has increased fuel abundance and changed forest structure, such fuel reductions treatments may realize ecological restoration goals. However, most fuel reduction treatments have been implemented in forest types that do not have high restoration needs (Schoennagel and Nelson, 2011), and treatment objectives may be driven entirely by social or economic concerns.

Piñon-juniper, or P-J, woodlands represent the third largest vegetation type in the continental U.S., occupying ca. 40 million ha (Laylock, 1999). They are highly variable and diverse, including particularly rich obligate bird communities. P-J woodlands contain the largest species list of nesting birds of any upland habitat type in the western U.S. (Balda and Masters, 1980), with up to 20% obligate species (Paulin et al., 1999) and many species of conservation concern (Colorado Partners in Flight, 2000; USFWS, 2008). For example, the Piñon Jay (*Gymnorhinus cyanocephalus*) has dropped by 4.27% annually in western forests since 1966 (Sauer et al., 2014). The Black-throated Gray Warbler (*Dendroica nigrescens*; -1.49%) and Virginia's Warbler (*Vermivora virginiae*; -1.37%) have also declined annually for the last 50 years (Sauer et al., 2014).

In spite of their ubiquity and ecological values, management of P-J woodlands has often been hindered by an inadequate understanding of their ecology, dynamics, and human impacts (Romme et al., 2009). Low-severity fire is not generally considered to have played an important role in shaping patterns of pre-settlement P-J woodland structure, where fire regimes were mostly characterized by rare stand-replacing fire (Romme et al., 2009). In many cases, direct management interventions such as thinning or fuel reductions may not represent ecological restoration (Baker and Shinneman, 2004). Management activities within P-J woodlands over the last ca. six decades have been primarily focused on tree removal to increase forage for wildlife and livestock (Aro, 1971; Redmond et al., 2014). These treatment techniques have recently been largely replaced by mastication (also referred to as chipping, shredding, or mulching) for fuel reduction and other habitat objectives. The immediate effect of mastication is to convert tree crowns into wood chips, in the process creating canopy openings, redistributing fuel from the canopy to the surface, converting large- to small-diameter woody fuels, and covering the ground with piles of woody debris and litter. Increased above- and below-ground resource availability is expected to boost the abundance and production of grasses, forbs, and shrubs. While mastication treatments are widely accepted as cost-effective means of meeting short-term fuel and vegetation management goals, there remains considerable uncertainty about the extent and duration of shifts in key ecological attributes and processes including native and non-native plant species composition, wildlife use, and fire behavior.

The purpose of our study was to assess the effects of fuel reduction treatment effects on avifauna, vegetation composition and structure, non-native plant species abundance, and surface and canopy fuel loadings in piñon-juniper woodlands of central Colorado. Between 1998 and the initiation of our study in 2012 the Bureau of Land Management Royal Gorge Field Office (BLM RGFO) conducted over 300 fuels reduction treatments on nearly 6900 ha of P-J woodlands, including 2800 ha of mastication treatments (Matt Rustand, Wildlife Biologist, BLM RGFO, personal communication). We contrasted response variables of interest in a series of treatment-control pairs that included 24 mastication and 5 hand-thin treatments. We conducted occupancy analysis of bird observation data at two scales. Local scale occupancy reflects the bird's habitat selection at the scale of its territory and represents direct habitat features selected by birds that would be altered by forest thinning treatments. Landscape scale occupancy reflects use of home range and provides a more regional understanding of bird distributional changes in relation to treatments. Because our study area embodied a chronosequence of 1-11-year-old treatments and a climate gradient, we also examined the effects of time-since-treatment and regional climate on birds, vegetation and fuels. Finally, we used our measured fuel data to develop custom models of fire behavior in treated and untreated stands under a range of fuel moisture scenarios, and we also simulated the effects of treatment modifications intended to increase fire resistance of residual trees.

Study Methods

Study Site

Our study encompassed piñon-juniper woodlands in the Arkansas River valley between Salida and Cañon City, Colorado (Figure 1), within lands managed by the BLM RGFO. Elevation across the study area ranges from 1,600 m to over 3,900 m; the elevation of Salida is 2,160 m and the elevation of Cañon City is 1,615 m. Piñon-juniper woodlands occupy upland sites at elevations < ca. 2,500 m. The BLM RGFO study area embodies a climate gradient: cooler and drier in the west and warmer and wetter in the east. Within our study area, we identified 24 different mastication and 5 hand-thin treatment units, each paired with an adjacent, untreated control unit, for field measurements of birds, vegetation, and fuels. Unit selection criteria included correct vegetation type (P-J woodlands), adequate area for four sample plots separated by >200 m, and accessibility. The treatments we examined were implemented between 2003 and 2014. Within each unit, we installed at random four sample plots separated by >200 m, for a total of 232 sample plots. Plot center locations (UTMs; NAD 83) were marked with survey stakes and recorded using a handheld GPS unit. At each plot, we conducted bird point counts and measured vegetation and fuels.

Field Measurements

We conducted 10-minute point counts at each point count station during each of three sessions in 2014 and 2015: May 15-31 (1st session), June 1-15 (2nd session), and June 16-30 (3rd session). Sampling was restricted to a five- hour time window (0500-1000 hr) each morning. During sampling, every bird that was seen or heard was identified and recorded, along with type of detection (visual vs. aural). We also indicated whether birds were located outside the intended sampling area and treatment or control site (for example, flying over, in adjacent agricultural fields, or other habitat types). Weather conditions were recorded, including wind speed, sky conditions, and air temperature.

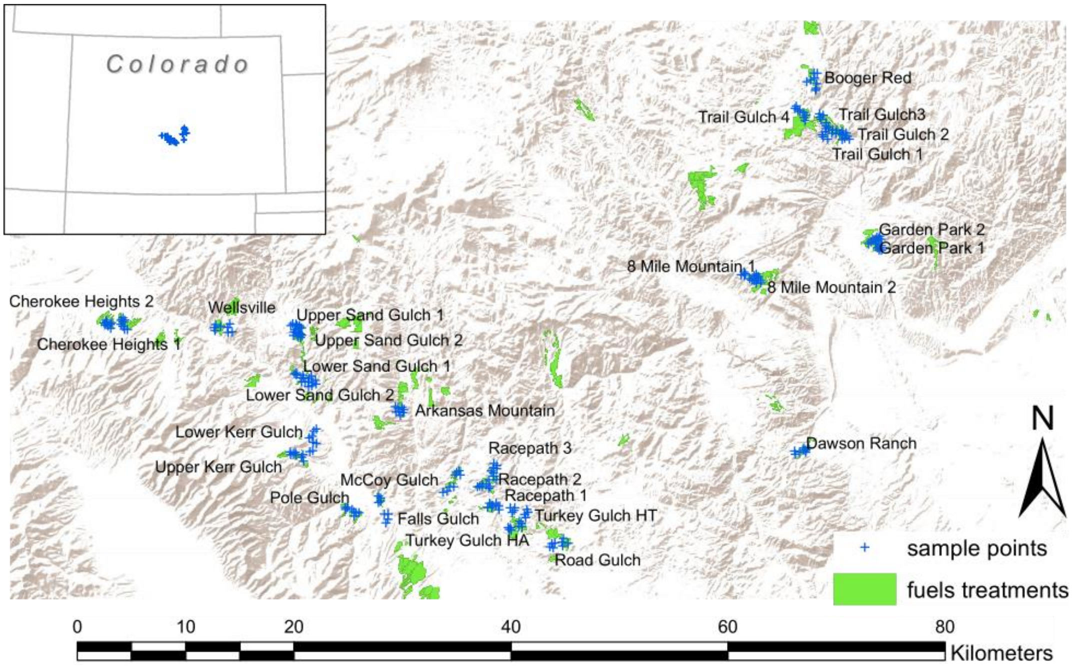

Figure 1. Locations of sample sites in hand-thin and mastication treatments and adjacent, untreated units in the Arkansas River valley of central Colorado, USA.

At each sample plot center we measured slope inclination, aspect, and elevation, and established a 5.64-m radius (0.01-ha) plot to measure the basal area and height, and visually estimate canopy cover, by species, for of all live and dead trees > 1.37 m height. We measured the live crown base height for all trees in each sample, and for treeless treated samples, from the nearest tree. Vegetation was sampled at 102 points spaced at 0.3-m intervals along three, 10.2-m line-point-intercept transects originating at the plot center. The first transect bearing was assigned at random, the second and third were offset by 120° and 240°, respectively. We recorded the species identity of all herbs, shrubs, and tree seedlings <1.37-m height, the ground cover type (cryptograms, bare soil, rock, litter, wood, or live plant), and the height of highest plant or down woody fuel contact with each intercept point ("hits") on each transect. We also noted whether points occurred beneath a tree (>1.37 m) canopy, tree species, and whether or not points occurred above a mastication debris (wood chip) pile. Surface fuels were also sampled by timelag moisture classes using Brown's planar intersect method (Brown, 1974) along three separate, 10-m transects separated by 120°. We recorded 1000-hour fuels by decay class along the entire transect, 100-hour fuels along the last five meters, and 1- and 10-hour fuels along the last two meters. At every meter (1, 2, …, 10) we recorded the depth of litter, duff, and any down wood, by size class. We also measured slope inclination of each transect and took a digital photo looking from the end toward plot center. Mean annual temperature (MAT) and mean annual precipitation (MAP) were extracted from 2-km resolution interpolated climate data (http://prism.oregonstate.edu); Heat Load Index (HLI) and elevation were extracted from 10-m digital elevation models; forest cover (at 10- ha and 100- ha scales) was estimated using GIS within 10- ha and 100- ha circles centered on each sample point using a supervised classification of 2012 and 2014 NAIP aerial imagery (Feature Analyst; http://www.textronsystems.com/what-we-do/geospatial-solutions/feature-analyst).

Analysis

We estimated multi-scale occupancy (MacKenzie et al., 2006; Nichols et al., 2008) for 31 bird species in the piñon-juniper bird community using a modified hierarchical approach (Pavlacky et al., 2012). We developed a three- step process to estimate multi-scale occupancy for each bird species which that included: 1) modeling detection probabilities (p) using model selection to determine the top detection model, 2) determining the most explanatory treatment effect model on avian occupancy at local (θ = theta) and landscape (Ψ = psi) scales, and finally 3) modeling covariate effects on landscape and local scale occupancy. Model development and analysis was done using Program Mark 8.0 (http://www.cnr.colostate.edu/~gwhite/mark/mark.htm; White and Burnham, 1999). In addition to avian point count data, we included 7 landscape scale covariates (year-since-treatment, mean annual temperature, mean annual precipitation, elevation, heat load index, forest cover at 10-ha scale and forest cover at 100-ha scale) and 11 local scale covariates (bare ground, herbaceous cover, shrub cover, vegetation height, standard deviation of vegetation height, number of stems of live trees, number of stems of live juniper, number of stems of piñon pine, basal area of live trees, basal area of junipers, and basal area of piñon pine).

To convert number of intersections of down woody fuels to fuel mass at each sample site, we used the composite (multi-species) equations of Brown (1974). Because the planar intersect method may yield low estimates of surface fuels in mulched fuelbeds, we also used the predictive equations developed by Battaglia et al. (2010) for P-J masticated fuelbeds to estimate total mulch fuel load (litter, duff, 1- and 10-hour fuels) within treatments, though estimates generated from these treatment-specific equations cannot be directly compared to those from untreated sites. Herbaceous and live woody surface fuels (shrubs) volumes were calculated from the vegetation point-line intercept transects (percent cover x height) and multiplied by 0.8 kg m^{-3} for herbs and 1.8 kg m^{-3} for shrubs. Fuelbed depth was calculated as the mean litter depth plus mean top "hit" height of any plant or down woody fuel. Canopy base height (CBH) was calculated as the mean crown base height (Sando and Wick, 1972; Ruiz-González and Álvarez-González, 2011) of all trees in each sample site. Canopy bulk density (CBD) was calculated from allometric equations fitted to each sampled tree by species at each site (Brown, 1978; Grier et al., 1992; Weaver and Lund, 1982). We followed Linn et al. (2013) in calculating CBD of piñon-juniper woodlands as the total of canopy foliage + all twigs < 2 mm diameter.

To characterize general patterns of plant community composition in untreated and masticated sites, and their associations with shifts in canopy cover, ground cover, topography, and climate, we conducted a non-metric multidimensional scaling (NMS) using R package 'vegan' (Oksanen et al., 2016). This analysis utilized coverages of 142 plant taxa averaged across all 12 transects at each of 48 sites (24 mastication treatments and 24 corresponding controls); cover was calculated as the total number of "hits" on the three point-line-intercept transects divided by the number of sample points (n = 102). We examined the relationships between ordination axes and treatment, cover by plant life form types (trees, shrubs, forbs, and graminoids) and species, ground cover, topography (elevation, slope, and heat load index), and regional climate gradients. To assess treatment impacts on non-native plant species we used paired t-tests to test for differing richness and total cover, and a Wilcoxon signed rank test to test for differing occurrence, in treatment vs.

control sample units. We also tested for differences in richness of native plants between controls and treatments. To test the relative importance of a) changes in ground cover (wood chip piles), b) removal of tree canopy, and/or c) disturbance and dispersal during the treatment process of these effects, we used generalized linear mixed effects models. We used paired t-tests to assess differences between control and mastication treatment units in surface and canopy fuels, including litter + duff, 1 + 10-hour fuels, 100 + 1000 hour fuels, herbaceous and woody (shrubs) surface fuels, fuelbed depth, tree cover, canopy bulk density, and canopy base height. Likewise, we tested for differences in the density and basal area of piñon, juniper (combined *J. monosperma* and *J. scopulorum*), and total tree seedlings. Tests were conducted at the unit-level, using the mean of four sample plots from each of the 24 pairs of units.

Fire Behavior Models

We conducted a principle component analysis (PCA) of 10 fuel variables to represent a gradient of surface and canopy fuels running from untreated to treated sites. The first two axes of our PCA accounted for 44% of the total variance of ten fuels parameters (25% on PCA 1 and 19% on PCA 2). The 1st and 2nd dimensions of the PCA of fuels parameters accounted for 43% of the variance in all 10 parameters. Canopy fuels declined and herbaceous fuels increased along PCA 1. Woody surface fuels increased along PCA 2. Abundances of PJ obligate birds also decreased along PCA 1 and 2, related to their close associations with PJ canopy. From these two dimensions, we generated a 9 x 11 grid (99 total points) and calculated fuel loads for each parameter at each point; these fuel loads were then used to create custom fuel models in BehavePlus. Initial model parameterization was based on the fuel model SB2 (Logging Slash Moderate Load) values, custom values were entered to represent the gradient from untreated to treated sites. In addition to the field-based estimates of values, we tested for the effects of two hypothetical additional changes: 1) pruning (an increase in canopy base height to 1 m), and 2) Rx fire (a 50% reduction in 1- and 10-hour surface fuels, and a 30% reduction in fuelbed depth). We used FireFamily Plus fire-weather associations to derive three fuel moisture and weather scenarios from two Remote Automatic Weather Stations (RAWS) within our study area: Four Mile (38° 32' 26", 105° 12' 14", 1923 m) and Copper Gulch (38° 18' 50", 105° 29' 04", 2365 m). Two scenarios representing 80th and 97th percentile conditions for the period 2011-2015 were used for modeling.

Key Findings

Treatment Effects on P-J Birds

Overall, treatments had strongly negative and persistent impacts on native, obligate avifauna. Nine species including Virginia's Warbler and Gray Flycatcher (piñon-juniper obligates), Mountain Chickadee, White-breasted Nuthatch (*Sitta carolinensis*), Steller's Jay (*Cyanocitta stelleri*), and Townsend's Solitaire (*Myadestes townsendii*; mature conifer species), Clark's Nutcracker (*Nucifraga Columbiana*) and Western Tanager (*Piranga ludoviciana*; open conifer species), and Northern Flicker (*Colpates auratus*; generalist) showed significant negative treatment effects at the landscape scale. In contrast, only 2 species responded positively to treatments at the landscape scale including the Piñon Jay and Mountain Bluebird (*Sialia currucoides*).

At the local spatial scale, 4 species showed significant negative treatment effects, and 4 species responded positively to local scale habitat changes. The Black-headed Grosbeak (*Pheucticus melanocephalus*) and Broad-tailed Hummingbird (*Selasphorus platycercus*; generalists), the Ash-throated Flycatcher (*Myiarchus cinerascens*; open conifer woodlands), and the Piñon Jay had greater occupancy in the control sites compared to treatments. The Lark Sparrow (*Chondestes grammacus;* edge species) occupied 20% of control sites at the local scale compared to 11% of hand thinned sites and 62% of mastication sites. The American Robin (*Turdus migratorius*) and Western Bluebird (*Sialia Mexicana*; open conifer woodlands), and Blue-gray Gnatcatcher (*Polioptila caerulea*; shrublands/woodlands) also had elevated local occupancy in treatments.

One species' occupancy responses were mixed at landscape and local scales. Piñon Jay occupancy increased in response to landscape scale treatments, but decreased at the local scale. Two species showed mixed responses to mastication and hand-thinning treatments: the Ash-throated Flycatcher (*Myiarchus cinerascens*) responded negatively to hand-thinning and neutrally to mastication, and the Mountain Bluebird responded negatively to hand-thinning but positively to mastication.

Treatment Effects on Vegetation and Fuels

Treatments drove major, persistent shifts in forest structure and composition, and canopy and surface fuels, relative to controls. Tree cover and canopy fuels were strongly reduced; concomitantly, down woody surface fuels (1- and 10-hour fuels) and live herbaceous fuels were elevated. Live woody (shrubby) surface fuels were not significantly different between controls and treatments, but increased significantly as a function of time-since-treatment, indicating increasing importance over time. Not only was the density and basal area of piñon and juniper strongly reduced in treatments, but it did not show any recovery with time-since-treatment. Likewise, seedling density was lower in treatments than controls, and did not show any increase over time, indicating that treatment effects will be persistent over many decades.

Across all samples, we identified 20 exotic species classified as introduced (USDA, NRCS 2015), including: crested wheatgrass (*Agropyron cristatum*), burdock (*Arctium minus*), cheatgrass (*Bromus tectorum*), littlepod false flax (*Camelina microcarpa*), musk thistle (*Carduus nutans*), Canada thistle (*Cirsium arvense*), bull thistle (*Cirsium vulgare*), bindweed (*Convolvulus arvensis*), stork's bill (*Erodium cicutarium*), prickly lettuce (*Lactuca serriola*), black medic (*Medicago lupulina*), alfalfa (*Medicago sativa*), yellow sweetclover (*Melilotus officinalis*), prickly Russian thistle (*Salsola tragus*), tall tumblemustard (*Sisymbrium altissimum*), salsify (*Tragopogon dubius*), puncturevine (*Tribulus terrestris*), white clover (*Trifolium repens*), mullein (*Verbascum thapsus*), and hairy vetch (*Vicia villosa*). Exotic species were much more frequently encountered at treated than control sites, occurring at 86% of sample plots in treatments and 51% of untreated sample plots (Wilcoxon signed rank test $P < 0.001$, 23 d.f.; Figure 2a). Richness of exotic species in treatments was more than double that of controls: 2.8 species in treatments, 1.3 species in controls (paired t-test $P < 0.001$, 23 d.f.; Figure 2b). Total cover by all exotics was much greater in treated (4.7%) than untreated sites (0.3%, paired t-test $P = 0.02$, 23 d.f.; Figure 2c). In contrast, richness of native species was nearly identical between treatments and controls (24.3 vs. 23.5, paired t-test, $P = 0.48$, 23 d.f.).

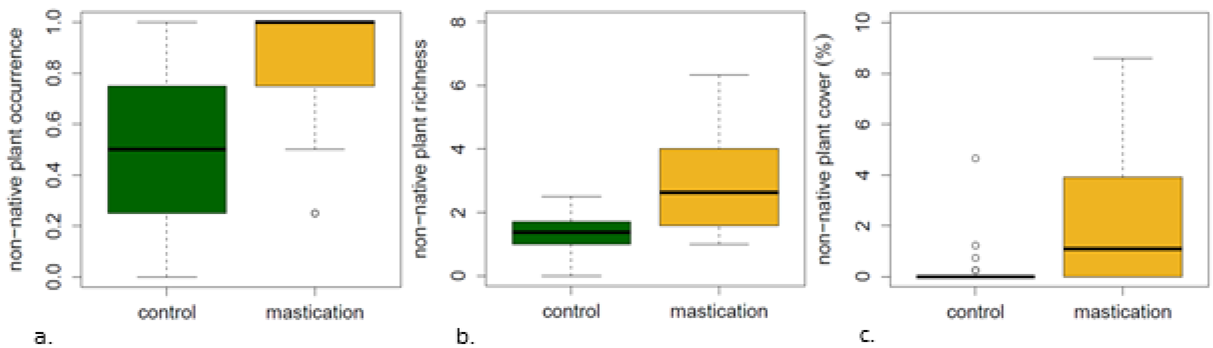

Figure 2. Non-native plant species a) occurrence in sample sites within each unit, b) richness, and c) cover in untreated controls vs. mastication treatments.

We found evidence for impacts of ground cover alterations, removal of canopy cover, and direct disturbance impacts of treatments on the abundance of exotic plants. Non-native species collectively and cheatgrass both showed a negative effect of tree canopy (positive effect of canopy removal) and a negative effect of wood chip piles. Cheatgrass also showed a positive effect of proximity (within 1 m) to wood chip piles. All non-native species were negatively related to bare soil. Finally, models of all annual non-native forbs and all non-native species showed strong positive effects of treatments that were not accounted for by tree canopy or ground cover variables.

Modeled Fire Behavior

Under the 80th and 97th percentile fuel moisture conditions, fuels treatments did well at reducing the risk of active crown fire (Figure 3). However, most treatments reduced crown fuels more than needed to change fire behavior. Measured canopy cover averaged 6.2% across all treatments, and 28.7% in controls. To reduce the active ratio < 1, canopy cover < 30-50% was sufficient under most conditions at the 80th percentile scenario, and canopy cover <15-35% was sufficient at the 97th percentile scenario. However, torching in these scenarios would still result in high tree mortality across most of the range of measured fuel loads (Figure 3).

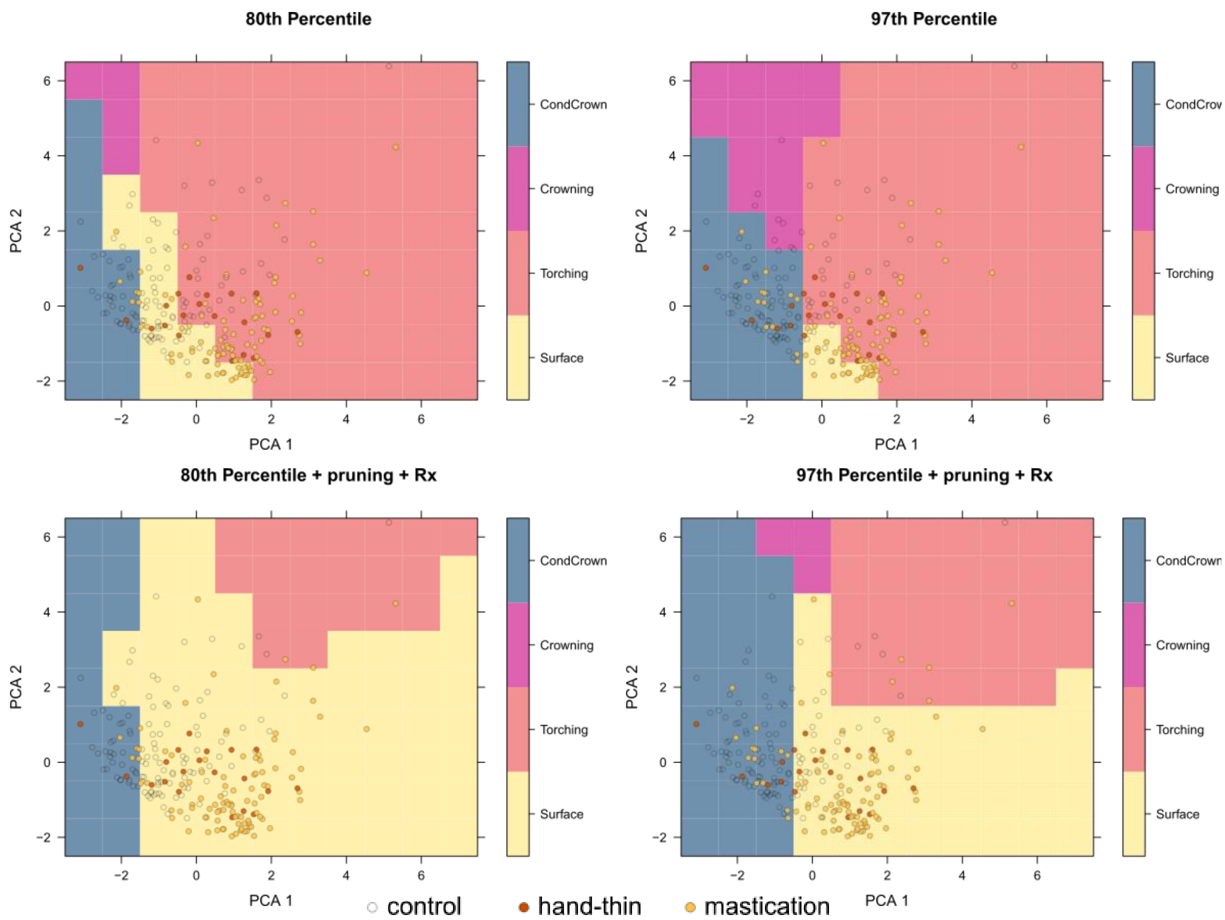

Figure 3. BehavePlus modeled fire types across the ranges of sampled fuel loads (PCA axes 1 and 2) under two different moisture scenarios. The top row represents actual conditions, the bottom represents outputs under simulated treatment modifications that reduced surface fuels and elevated canopy base height.

Most treatments in P-J woodlands, particularly mastication treatments, are aimed at reducing canopy fuels, thereby lowering ***active ratios***. However, we also simulated two factors aimed at decreasing ***transition ratios***, via increasing the canopy base height (for example, by pruning) and reducing surface fuels (for example, via Rx fire). Both simulations led to major reductions in transition ratios and thus the risk of crown fire.

Management Implications

Acknowledge Trade-offs

Fuels reductions and other thinning treatments are frequently presented as win-win interventions that generate ecological and social benefits. In some settings such as formerly frequent-fire forest types, treatments may restore historic ecological conditions and processes and also reduce fire hazards to human values at risk. However, in many other settings such as persistent P-J woodlands, fuel treatments may not restore lost ecological functions, and instead may move systems away from reference conditions (Baker and Shinneman, 2004; Romme et al. 2009). Taken together, the changes in avian occupancy, vegetation structure, composition, and fuel loads imparted by treatments, particularly mastication, in our study area, are expected to have major and sustained influences on key ecological processes including habitat use by a range of species and fire in P-J woodlands. Treatments are certainly highly effective in reducing the risk of active crown fire, generating a mosaic of different habitat types within otherwise continuous woodlands, and yielding substantial increases in the cover and biomass of grasses and forbs, particularly at the warmer and more mesic end of the climate gradient in our study area. However, we also identified several potential undesirable consequences of mastication in P-J woodlands including major expansions of non-native plant species and changes to fire behavior associated with more abundant and likely drier surface fuels. Under some circumstances (for example, within a densely populated WUI), these consequences may be acceptable to reduce fire hazard. Under others (for example, far from human values at risk), they may not. We caution managers to implement treatments prudently in the light of persistent, negative ecological impacts that accompany woodland thinning in PJ ecosystems.

Are Treatments Necessary?

We also encourage managers to critically consider whether or not treatments may even be necessary in any given setting. Nationally, a high proportion of fuels reduction treatments take place in locations distant from the Wildland Urban Interface (WUI) and in forest types that require no ecological restoration (Schoennagel and Nelson, 2011). A management culture focused on cost-effectiveness and treatment quantity (annual objectives measured in acres treated) may conflict with more judicious, but higher quality interventions (e.g., Lehmkuhl et al., 2007). Finally, while mastication is a modern approach to forest treatments, it also represents the continuation a many-decades-old management paradigm focused on reducing P-J woodlands for a host of social, economic, and ecological reasons. Projected impacts of climate change present questions to this management paradigm. Recent hot drought linked with episodic bark beetle (*Ips* sp.) outbreaks have driven recent, region-wide die-offs in P-J woodlands (Breshears et al., 2005), and ongoing climate change is projected to drive unprecedented additional tree mortality over the next century (McDowell et al., 2016). The extent to which thinning treatments may increase the resistance of residual trees to future drought is not known, and should certainly be the subject of future research. However, it is unlikely that treatments will promote resistance to coupled, large-scale insect outbreaks. As such, anticipated landscape-scale woodland sparsification and regional contraction of P-J woodlands call into question whether limited management resources are best invested in tree removal projects in these systems.

Mitigate Non-native Plant Species

Disproportionately large increases in the abundances of non-native plant species in treatments relative to increases in occurrence and richness (at least one non-native species occurred in at least half of all control plots) suggest two different explanations: 1) expansion of propagule sources in treatments may have promoted colonization of nearby untreated stands, and/or 2) presence of these species across the pre-treatment landscape at low levels led to rapid expansions following suitable disturbances in treatments. Further research may be useful to disentangle these different explanations, as they lead to different management implications. If treatments are also likely to increase the presence of non-native species in adjacent untreated stands, further precaution is warranted on the part of managers in prescribing treatments, particularly in relatively pristine areas lacking these species. Best management practices to prevent introduction, including washing machinery to remove seeds prior to site entry, will continue to be valuable, but post-treatment control measures might also be considered. The second explanation suggests increased caution is needed in areas already containing a rich non-native species pool, where treatments may trigger population expansions. Preventing introduction by cleaning machinery may not be useful if seed sources on site are subsequently dispersed throughout treatments units. Here, managers might consider proactive, pre-treatment weed control measures. Other authors have recommended seeding treatments with native species (Redmond et al., 2014; Young et al., 2013). Increased competition with natives, particularly at early stages of invasion, may be helpful, but the potential for contaminated seed mixes also poses risks. In addition to pre- and post-treatment weed control measures, piles of mastication debris could be redistributed across treatment units to reduce bare soil cover and eliminate microenvironments conducive to cheatgrass colonization adjacent to mulch piles.

Retain More Trees

Modeled fire behavior suggests that under most conditions, treatments will be highly effective at reducing active crown fire risk, but also that treatments generally removed more trees than necessary to mitigate this risk. Under moderate burning conditions (80^{th} percentile), model outputs suggest that most untreated stands could not support active crown fire. Even under extreme burning conditions (97^{th} percentile), stands with canopy cover at 15% did not support active crown fire. In contrast, treatments averaged only 6% canopy cover. While these model outputs should be viewed as hypotheses that require further empirical testing, they suggest that managers may be able to acceptably reduce fire hazard in most settings with less aggressive treatments. Retention of more trees within treatments will benefit P-J obligate birds and may also result in less expansion by non-native plant species. We also encourage managers to experiment with different treatment configurations that retain larger islands of intact habitat, rather than uniformly spaced but highly artificial P-J “savannas”.

Follow-up Measures to Increase Fire Resistance

Decreased canopy fuels and increased herbaceous surface fuels including exotic annuals are expected to alter potential fire behavior via reduced active crown fire probability, but also increased surface fire intensity, flame length, and rate of spread. Fire behavior models indicated that most residual trees were still highly susceptible to passive crown fire (torching), particularly in grassy and masticated sites with high surface fuel loads. Models including simulated treatment modifications suggest that this remaining crown fire risk could mostly be eliminated via follow-up surface fuel reductions (e.g., Rx fire) and/or crown base height increases (e.g., pruning). As such, we propose that managers consider additional post-treatment fuels interventions to increase the landscape-scale fire resistance, which is anticipated to yield both social and ecological benefit. Reductions of woody surface fuels might be achieved through post-treatment, prescribed burning, though this may promote additional expansion by non-native species, and additional monitoring and research are needed. We also recommend managers experiment with follow-up pruning of low tree branches to increase crown base height and fire-resistance of residual trees, though we recognize that this may not always be achievable due to woodland tree morphology (e.g., junipers with multiple stems and many low branches).

Relationship to Other Work

Thinning treatments in P-J woodlands, particularly mastication, may represent no-analog disturbances, and despite a growing body of literature (Battaglia et al., 2010; Fornwalt et al., 2016; Huffman et al., 2009, 2013; Owen et al., 2009; Roundy et al., 2014; Young et al., 2013) there remain considerable uncertainties with regards to ecological impacts. In this project, we identified several potential undesirable consequences of thinning treatments in P-J woodlands including reductions in the occupancy of native, obligate bird species, major expansions of non-native plant species, and changes to fire behavior associated with more abundant surface fuels.

Several previous studies have showed negative effects of piñon-juniper tree removal (type conversion via chaining) on birds both in the short- (O'Meara et al., 1981; Sedgwick and Ryder, 1986; Crow and van Riper, 2010) and long-term (Gallo and Pejchar, 2017), however, tree thinning treatments often had negligible effects on birds with the exception of woodland functional groups (Bombaci and Pejchar, 2016). Conifer woodland obligates including piñon-juniper specialists show the strongest negative responses to treatments; 10 of 14 conifer obligate species (71%) declined at one of the 2 scales we considered (landscape and local). In our study, 6 conifer obligates exhibited strong negative effects of thinning and 9 of 24 species (38%) analyzed in this study (where occupancy parameters were estimated) negatively responded to thinning at the landscape scale, suggesting that woodland reduction treatments have the potential to affect regional distributions and populations of forest birds (Pavlacky et al., 2012). The other 4 species that declined in treatments were impacted at the local scale, indicating that thinning treatments may reduce the number of suitable territories in highly managed areas. Tree thinning treatments reduced canopy cover and tree density, thus impairing habitat suitability for forest obligate species, but simultaneously enhanced habitat for 6 bird species (25%). The strongest positive responses came from the Mountain Bluebird at the landscape scale and the Lark Sparrow at the local scale. Both species strongly associate with habitat ecotones (Power and Lombardo, 1996; Martin and Parrish, 2000), habitat structure boosted by mastication and hand thinning treatments. Piñon Jays showed a mixed response to treatments. At the territory scale, occupancy declined on treated sites. However, occupancy increased for Piñon Jays in treatments at the landscape scale. Piñon Jays generally nest and roost in dense stands of primarily piñon pine within 800 m of an edge, and forage and cache in more open landscapes up to 6 km from the forest boundary (Ammon and Boone, 2014; Kristine Johnson, personal communication). Thus, Piñon Jays may find treated landscapes suitable for occupancy as long as they contain fairly dense patches, but within treated forest stands, fragmentation may exceed a threshold that Piñon Jays can tolerate for nest and roost habitat selection.

Substantial, persistent canopy reductions coupled with sparse tree regeneration suggest mastication treatments in P-J woodlands are likely to have an effective duration of many decades. Persistent treatment effects in these relatively unproductive P-J woodlands is consistent with delayed responses to other disturbances such as historic fire (Barney and Frischknecht, 1974) or chaining treatments (Redmond et al., 2013), but contrasts with rapid tree regrowth following mastication treatments in more productive forest types (e.g., Stephens et al., 2012). We also found increases in herb and shrub cover as a function of time-since-treatment across our 11-year chronosequence. Similarly, Fornwalt et al. (2016) reported increases in grasses and forbs in mulched P-J woodlands continuing through at least 6-9 years post treatment.

Three major shifts in vegetation composition associated with mastication treatments included 1) reduced cover by both piñon and juniper trees, 2) increases by a suite of shade-intolerant, native grasses, and 3) a considerable expansion of weedy, non-native species. Each of these shifts also showed relationships to measured climate variables across our study area. Though we did not find significant shifts in the relative proportions of piñon and juniper trees or seedlings in treatments vs. controls, the slight proportional decreases in piñon and increases in juniper are suggestive of longer-term changes reported by others (Redmond et al., 2014; Schott and Pieper, 1987) towards increased juniper dominance following thinning. Masticated treatments in our study area did not yield changes to native species richness, in contrast with findings elsewhere (e.g., Fornwalt et al. 2016), though patterns of cover by these species changed substantially. As has been found following P-J tree removal treatments elsewhere in the western U.S. (Fornwalt et al. 2016; Huffman et al., 2013; Ross et al., 2012; Roundy et al., 2014), we found substantial increases by a suite of native, perennial P-J understory species. It might be expected that these increases will persist until tree canopy cover rebounds. However, Schott and Pieper (1987) reported that grass cover fell back to pre-treatment levels in ca. 25 years following woodland removal treatments in New Mexico.

Numerous other studies have reported increases in the abundance of non-native plant species, particularly cheatgrass, in P-J thinning treatments (Huffman et al., 2013; Owen et al., 2009; Redmond et al., 2014; Ross et al., 2012; Roundy et al., 2014; Stephens et al., 2016; Young et al., 2013). Patterns of occurrence and richness also point toward large expansions into mastication treatments, but additionally indicate a large regional species pool across both treated and untreated landscapes. Predictive models of non-native plant species occurrence suggest that increases are driven by canopy reduction, changes to the soil surface, and other unsampled factors imparted by treatments. Models for cheatgrass alone and all non-native species together indicate strong negative associations with tree canopies, indicating that increased light availability (or perhaps below-ground resources such as moisture or nitrogen; Ross et al. 2012) enhanced colonization and growth in treatments. The effects of mastication debris piles in suppressing herbaceous plant growth have been reported in other forest types (Wolk and Rocca, 2009), and appear to similarly reduce cheatgrass colonization in our study area. Three of the non-native herbaceous species that increased in treatments, cheatgrass and the two tumbleweeds (prickly Russian thistle and tall tumblemustard) are expected to produce changes in fire behavior and post-fire recovery trajectories (Zouhar et al., 2008). In some systems, cheatgrass establishment and expansion alters fire cycles by changing fuel conditions and creating positive feedbacks that promote further expansion (Brooks et al., 2004; D'Antonio and Vitousek, 1992). Recent increases in burning in P-J systems in some settings has been attributed to cheatgrass expansion (Arendt and Baker, 2013; Miller and Tausch, 2001.

Our findings point towards large potential shifts in fire behavior associated with mastication treatments in P-J woodlands. First, active crown fire risk in treatments is expected to be essentially zero. Not surprisingly we found that treatments effectively produced substantial and persistent reductions in canopy fuels. Active crown fire is rare in untreated P-J woodlands except under extreme burning conditions (Romme et al., 2009 and references therein), due to generally low canopy fuel loads (i.e., canopy bulk density averaged 0.214 kg m^{-3} across all our untreated control sites). However, treatments also led to major changes in surface fuel abundances and characteristics. We found pronounced increases in down woody fuels (mastication debris), though these showed gradual declines with time-since-treatment. Standard methods (e.g., Brown, 1974) for estimating down woody surface fuels are known to underestimate their abundance in masticated fuelbeds, where deep and compact mulch piles may contain considerably more fuel than surface-based estimates would indicate, and we found considerable differences between our estimates of down woody fuels derived from Brown's (1974) equations versus those provided by Battaglia et al. (2010). However, spreading surface fires, as modeled via Rothermel's (1972) algorithms, are unlikely to consume deep mulch piles, which are prone to prolonged smoldering (Knapp et al., 2012), and as such, standard and superficial estimates of only the top layer of these fuels may lead to more accurate predictions of fire behavior. Collectively, increases in surface fuels are expected to lead to increased potential for surface fires via increased fuel continuity, increased surface fireline intensity, flame length, and rate of spread (e.g., Rothermel, 1972), particularly where P-J mastication treatments are conducted in topographic and climatic settings that promote abundant grass growth, especially of annual grasses such as cheatgrass. Our findings also suggest that treatments may not consistently lead to more fire-resistant stands of residual trees. Knapp et al. (2012) examined mastication effects on prescribed fire behavior in a ponderosa pine forests, and found that crown scorch still imposed high mortality among residual trees in mastication treatments, particularly under extreme fire weather conditions. This is likely to be even more true in P-J woodlands, where crown base heights are low (0.35 m in sampled treated sites), and fires generally only burn under extreme conditions.

Future Work Needed

Our research examined the effects of fuels treatments in P-J woodlands on birds, vegetation, fuels, and modeled fire behavior. Each of these aspects suggests avenues for further research.

While we found strong negative effects on occupancy of P-J obligate and forest-dependent avifauana, our study did not directly address abundances of these species in treated vs. untreated areas, nor patterns of habitat use, or reproductive success or other metrics of performance. Occupancy is a conservative measure, and the differences in occupancy we found were small in comparison with differences in the number of observations of different species. Treatment impacts on populations of P-J obligates are likely to be much greater than impacts on occupancy, and quantifying these impacts may be useful to guide management decisions, especially if populations continue to decline for many of these species. Likewise, our study did not characterize how birds may utilize these habitats differently (e.g., foraging, nesting, etc.), which may also be important. This is particularly true for the Piñon Jay, whose occupancy increased in response to landscape scale treatments, but decreased at the local scale, suggesting divergent scale-dependent habitat selection. We also suggest that a better understanding of treatment effects on piñon cone crops could be useful for understanding impacts to Piñon Jays and other species that depend on piñon pine nuts, such as the piñon mouse. While treatments reduce the abundance of piñon trees across large landscapes, there is reason to expect positive impacts on the cone crops of residual trees, which could partially offset reductions. If reduced competition in treatments increases cone crops, this would also suggest that treatments could be optimized for Piñon Jays (e.g., thinning to a particular density, and increasing the fire resistance of residual trees). Beyond birds, the impacts of treatments to a wide range of biota remain largely unknown, most importantly, whether intended beneficiary species (e.g., sage-grouse, mule deer, etc.) actually show population increases or expanded habitat use post-treatment.

While our study examined treatment impacts across a local climate gradient and across a 1-11 year chronosequence, the extent and duration of treatment impacts, and their relationship to climate, remains an area of considerable uncertainty. Treatments are likely to have much more lasting impacts within parts of the geographic range and topoclimate context of piñon-juniper woodlands than others, and contrasting treatment impacts across the climatic range of this forest type may lead to important insights. In particular, whether or not treatments might promote greater resistance to drought due to reduced competition will become an increasingly important question. Within our study area there is anecdotal evidence that piñon pine trees in treatments were still highly vulnerable to a recent drought-related *Ips* outbreak. Effective means for controlling or reducing unintended non-native plant expansions in treatments should also be a high priority for further research.

Very little is known about actual consequences of P-J treatments on fire behavior and effects, particularly in mastication treatments. Empirical work is certainly called for, in the form of experimental studies, and as treatments increase across the landscapes and are exposed to wildfire, examination of fire effects.

Deliverables Crosswalk Table

Deliverable Type	**Status**	**Delivery Dates**
Peer-reviewed publications	Currently, one manuscript on fuels treatment effects on vegetation and fuels is in review at Forest Ecology and Management. A second manuscript on effects to bird communities is in an internal (Colorado Parks and Wildlife) review process, and will be submitted to the journal Condor in early 2017. A third manuscript on modeled fire behavior in P-J treatments is in preparation.	2016 & 2017
Field Trip(s)	With the SRFSN, we coordinated a one-day field trip to visit a number of treated sites in the BLM RGFO in June 2016. Over 50 researchers and land managers participated.	2016

Webinar(s)	To reach out to managers outside of our area (the southern Rockies), we have planned to deliver a webinar coordinated by the Southwest Fire Science Consortium in 2017.	2017
Fact Sheet(s)	In lieu of a fact sheet, we worked with the SRFSN to develop a short video describing our study findings, hosted on the SRFSN youtube channel, https://youtu.be/azimblL3O5Y.	2016
Presentations	Study findings have been presented at a total of ten invited or contributed oral presentations and one poster presentation to researchers, managers, and other stakeholders. Presentations that specifically targeted managers include the Association for Fire Ecology (in 2015 and 2016), Society for American Foresters Colorado and Wyoming meeting (2016), the New Mexico P-J symposium (2016), and the Colorado Chapter of the Wildlife Society (2016).	2015 & 2016
Other	Study data and metadata have been sent to the RMRS Data Archivist, and will be archived in early 2017.	2017

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