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The effects of thinning and burning on understory vegetation in North America: A meta-analysis

Joshua Willms [a], Anne Bartuszevige [b], Dylan W. Schwilk [a], Patricia L. Kennedy [c,*]

[a] Texas Tech University, Department of Biological Sciences, Lubbock, TX 79409-3131, United States
[b] Playa Lakes Joint Venture, Lafayette, CO 80026, United States
[c] Eastern Oregon Agriculture & Natural Resource Program and Department of Fisheries and Wildlife, Oregon State University, Union, OR 97883, United States

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ABSTRACT

Management in fire-prone ecosystems relies widely upon application of prescribed fire and/or fire-surrogate (e.g., forest thinning) treatments to maintain biodiversity and ecosystem function. The literature suggests fire and mechanical treatments proved more variable in their effects on understory vegetation as compared to their effects on stand structure. The growing body of work comparing fire and thinning effects on understory vegetation offers an opportunity to increase the generality of conclusions through meta-analysis. We conducted a meta-analysis to determine if there were consistent responses of understory vegetation to these treatments in North American forests that historically experienced frequent surface fire regimes (<20 years fire return interval, FRI). Means and standard errors were extracted from 32 papers containing data on the response of four understory functional groups (herbaceous, shrub, non-native, and total) to thinning and burning treatments to calculate effect sizes. Lack of replication and inconsistent reporting of results hindered our ability to include many studies in this analysis. For each response variable (species richness and percent cover), we compared three treatment pairs: burn vs control, thin vs control and thin vs burn. We calculated standardized mean differences (Hedges' g) for each pair and tested if this differed from zero using a random effects model fit with restricted maximum likelihood to account for variation by site. The most consistent effect of the treatments was the increase in non-native species following mechanical thinning and reduction in shrub cover following a burn. These differences suggest the two treatments may not be surrogates in the short-term (less than 5 years). Increase of non-native species due to disturbance is well established but it is not clear if burning and thinning consistently have differential impacts. Response of non-native plants to disturbance is likely a complex function of a variety of site and landscape factors that cannot be evaluated by the current literature. We conclude that prescribed fire and thinning treatments can be used successfully to restore understory species richness and cover, but they can create different conditions and these potentially different outcomes need to be considered in the planning of a fuels reduction treatment. We discuss management options to reduce negative effects of the treatments and we suggest managers use current decision-making frameworks prior to designing an intervention.

1. Introduction

North American frequent-fire forests have been shaped by fire over evolutionary and ecological time scales. However, for much of the 20th century, land managers concentrated on minimizing amount of land that burned. Compared to presettlement fire regimes in many contemporary forests, fire intervals have lengthened (Cyr et al., 2009; Aldrich et al., 2010; Spetich et al., 2011), although there is evidence for significant variability in historical fire return intervals (Odion et al., 2014). Increased recognition of the central role of fire in maintaining forest structure and function has contributed to a shift from fire exclusion to reintroduction of fire in fire-dependent forests, with the aim of reducing fuels and restoring historic stand structure (Agee and Skinner, 2005). This recognition has prompted U. S. federal initiatives such as the National Fire Plan and Healthy Forest Restoration Act (2003) that mandate federal land managers restore forest structure and function and reduce risk of wildfire on federal lands. Use of widespread (i.e., over a large area) fuel treatments has led to increasing discussion of the effectiveness, suitability and ecological impacts

* Corresponding author.
E-mail address: pat.kennedy@oregonstate.edu (P.L. Kennedy).

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of thinning and prescribed fire (Schoennagel et al., 2004; Schwilk et al., 2009; Schoennagel and Nelson, 2011; Stephens et al., 2012).

Although prescribed fire is often the preferred fuel reduction practice, forest managers often face social and economic constraints on burning particularly when human safety and property might be compromised. Additionally, in forests thought to have departed significantly from historical fire return intervals, there is concern that introducing fire may result in unnaturally high intensity fire that may be difficult to manage or may have negative ecological effects (but see Bond et al., 2012; Fontaine and Kennedy, 2012). Therefore, mechanical fuel reduction methods have increasingly been used to reduce fuels or restore historic stand structure (Crow and Perera, 2004). Uncertainty regarding the relative ecological effects of prescribed fire versus mechanical treatments has led to increasing attention on these so-called ''fire surrogates" such as the National Fire and Fire Surrogate study (Schwilk et al., 2009; McIver et al., 2013).

Early forest management emphasized recruiting trees for commercial harvest. However, in the last half of the 20th century, forest management practices shifted focus to include managing for ecosystem services, including biodiversity. In most forests, the majority of plant biodiversity is in the understory herbaceous layer. In addition to harboring high diversity, understory herbaceous communities have profound effects on other ecosystem services such as forest nutrient cycling (reviewed by Gilliam, 2007). Most attention has been paid to the effects of fire and mechanical treatments on forest structure and fuels (e.g., Moghaddas et al., 2008; van Mantgem et al., 2011; Kreye and Kobziar, 2015), and reviewed in Fulé et al. (2012); the extent to which mechanical treatments or thinning approximate effects of prescribed fire on forest understory vegetation is not as well understood. Results from the National Fire and Fire Surrogate study demonstrated that fire and mechanical treatments proved more variable in their effects on understory vegetation as compared to stand structure (Schwilk et al., 2009). This is not entirely surprising: although both fire and thinning remove overstory trees and allow increased light to reach understory plants, extent of canopy removal varies with thinning intensity and fire severity. According to a recent review by Abella and Springer (2015) treatments must reduce tree canopy cover to <30–50% to elicit appreciable responses from the forest understory.

In addition to variable treatment effects on forest cover, fire and thinning modify the abiotic environment differently. Fire restructures microsites and soils that many plants depend on for germination and growth (Bond and van Wilgen, 1996; Gundale et al., 2005, 2006; DeLuca et al., 2006). Thinning, on the other hand, removes or rearranges (as opposed to consumes) vegetation and may alter nutrient dynamics (e.g., Boerner et al., 2006). Many mechanical thinning methods also result in soil disturbance and compaction that fire does not cause (Schwilk et al., 2009). The differences exhibited between fire and fire surrogate treatments may result in differences in responses between native and nonnative species, and in the percent cover and species richness in the herbaceous and shrub layers (Dodson, 2004; Wienk et al., 2004; Metlen and Fiedler, 2006; Collins et al., 2007; Nelson et al., 2008; Zhang et al., 2008b; Fornwalt and Kaufmann, 2014). The growing body of work comparing fire and thinning effects on understory vegetation (recently reviewed by Abella and Springer, 2015) for mixed conifer forests in North America) offers an opportunity to increase the generality of conclusions through meta-analysis.

Our objective was to conduct a meta-analysis of the literature that investigated effects of thin and burn treatments on understory species in North American. Specifically, we were interested in the degree to which thin treatments mimic prescribed burn treatments, and to what extent burning or thinning differ from control treatments. We tested three pairwise comparisons: thinning treatments versus controls, burning treatments versus controls, and thinning treatments versus burning treatments for percent cover and species richness in total species, non-native species, herbaceous species, and shrub species.

We tested the following hypotheses: (1) total species richness and cover of herbaceous understory plants will increase in thin and burn treatments compared to controls as an effect of increasing light availability (Wienk et al., 2004; Metlen and Fiedler, 2006; Fornwalt and Kaufmann, 2014); (2) total cover of understory shrubs will decrease in response to burning, but not to thinning in the short term because burning consumes understory shrubs and these are slower to respond to increased light than are herbaceous species (Nelson et al., 2008; Zhang et al., 2008b); and (3) non-native plant species are often favored by disturbance and we expect both thinning and burning to increase non-native species richness and cover relative to controls with the greatest increases in thinning treatments as a result of greater soil disturbance in thinning relative to burning (Dodson, 2004; Collins et al., 2007).

2. Materials and methods

2.1. Literature search and vetting

In May 2014, we performed a search of the scientific literature investigating effects of prescribed fire and thinning treatments on understory vegetation. We used multiple databases: ISI Web of Science (http://www.webofknowledge.com) and AGRICOLA (http://agricola.nal.usda.gov/) both of which searched literature published since 1970 and Forest Science (http://www.cabi.org/forestscience/) which searched literature published since 1939. We also supplemented these searches with a Google Scholar search (http://scholar.google.com/) which, despite limitations in coverage, includes gray literature publications as well as proceedings. In addition to these search engines, we included additional references gleaned from publications found in the literature search.

We used the following search terms ('⁎' indicate wild card searches uses to include plural forms, etc.):

- Understory AND native*
- Percent Cover AND native*
- Fire AND Understory*
- Understory AND exotic*
- Percent Cover AND exotic*
- Fire AND Percent Cover*
- Understory AND forb*
- Percent Cover AND forb*
- Burn* AND Understory
- Understory AND graminoid*
- Percent Cover AND graminoid*
- Burn* AND Percent Cover
- Understory AND shrub*
- Percent Cover AND shrub*
- Thin* AND Understory
- Thin* AND Percent Cover.

The literature search from the databases yielded approximately 3500 references, which were then vetted for appropriate material. Documents were eliminated that dealt with medical issues (i.e., new treatments for burn victims), investigations of ecological processes related to fire but not relevant to the scope of this document (e.g., nutrient cycling, insect infestation), or modeling studies with little empirical data. Because North American studies comprise the bulk of the literature and our power to examine larger geographic patterns would be very low, we restricted our analyses to North America. We were specifically interested in studies that collected quantitative data on the response of understory plants to a prescribed fire or thinning treatment. We excluded papers that only reported the combined treatment of thinning and burning but did not include thinning or burning as separate treatments. This vetting process yielded 57 references.

Because statistical reporting was not uniform across references, we performed a second round of vetting to exclude papers that could not be placed in a quantitative meta-analysis. Papers that lacked replicated treatments or failed to include a measure of variation (standard error, standard deviation or variance) were excluded. In addition, studies that reported data collected 10 or more years after the thinning or burning event were excluded because there wasn't sufficient literature to quantitatively examine long-term effects of the treatments on understory vegetation. When multiple papers reported on the same study (determined by location and study dates), we used only one of the published papers. Varying levels of prescribed burn severity were inherent due to differences in species composition, terrain, weather, and season. There were insufficient studies to quantitatively evaluate effects of burn severity on understory vegetation. Thus, for studies that reported data for multiple levels of burning severity, the moderate level of burning was selected for inclusion in this analysis. Varying forms of thinning and mastication were used in thinning treatments. These included chainsaw, dragging a chain between tractors, hand-thinning, thin-and-pile, thin-and-scatter, thin-only, thin-and-chip, partial-cut, and clear-cut. Data were insufficient to quantitatively evaluate effects of various forms of thinning on understory vegetation. Thus, for each paper that reported several types of thinning, data were selected that were as close to a thin-only treatment as possible. If multiple thinning-only methods were used, we pooled data across methods when possible. In some cases, separate papers reported results from the same experiment (e.g., Phillips et al., 2007; Waldrop et al., 2008). In these cases, we did include both as separate studies if they reported different response variables but we avoided including duplicated sites in any particular contast.

2.2. Data extraction and analysis

We investigated effect of burning and thinning treatments on two response variables that describe effects on understory vegetation: species richness and plant cover. Each of these was recorded separately for each of two growth forms (herbaceous species and shrubs) and separately for non-native species and all species combined. This resulted in four vegetative categories (herbaceous, shrub, non-native and total) and two response types (cover and species richness) for a total of eight possible response variables, although not all were available in each study. Although the shrub and herbaceous categories are exclusive of one another, the other categories can overlap. For example, in several studies, non-native species are a subset of herbaceous species. When plots at multiple spatial scales were included in a study, we selected the scale closest to 100 m^2 as this was the most commonly reported scale across studies.

For each selected article, we extracted treatment means, sample size, and either standard deviations or standard errors of the mean for each response variable from the results text, tables or figures as required. In some cases, papers reported differences pre-post study and we used those differences rather than raw richness and cover values. We calculated pairwise treatment effect sizes for each response variable for three different pairwise comparisons: Thin vs Burn, Thin vs Control, and Burn vs Control. For each comparison, we calculated the bias-adjusted estimate of the standardized mean difference [“Hedge's g”, a bias corrected version of Cohen's d (Gurevitch and Hedges, 2001)] with the following equation:

$$g = \frac{X^E - X^C}{S^{EC}} \tag{1}$$

where X^E is the mean value of the response variable in the “experimental treatment” (Burn or Thin depending on contrast), X^C is the mean value of the response variable in the “experimental control” (Control or Burn depending on the contrast), S^{EC} is the pooled standard deviation of both groups, and J is a term that corrects for bias due to small sample size (Gurevitch and Hedges, 2001). The effect size, g, can be interpreted as the difference between the cover or species richness of plants in treatments relative to controls, measured in units of standard deviations.

We conducted all analyses in R (R Development Core Team, 2013) using the *metafor* package (Viechtbauer, 2010). We assumed that true effect sizes varied across sites (treatments differ in details, forests differ in absolute cover and richness, and plot sizes could vary) and therefore, we used a random-effects models (Gurevitch and Hedges, 2001). We parameterized models using restricted maximum-likelihood. To test whether mean effect sizes for a comparison differed significantly from zero, we assumed a normal distribution of effect sizes and their confidence intervals (Viechtbauer, 2010). Because of the large number of response variables and pairwise treatment comparisons, we used a Holm p-value adjustment (sequential Bonferroni correction) to control for multiple comparison error rate. Our *a priori* alpha-level was 0.1, which balanced the likelihoods of Type 1 and Type 2 errors.

A handful of assumptions were necessary to reconcile each reference into a single, comparable format. When papers reported mean species richness or percent cover values and standard errors for multiple years per treatment, we selected the data closest to the average years since treatment for the overall data set (3 years). Because native species represented over 99% of species richness and percent cover for papers in which both parameters were reported, ‘native’ was substituted for ‘total’ species in cases where papers did not separate native and non-native species (Collins et al., 2007; Dodson et al., 2007, and Huffman et al., 2013).

The final data used to conduct this analysis were reported from a wide array of locations, forest types, and ecological management histories (Table 1, Fig. 1). To account for this variation in sampling methodologies and time-lines the following covariates were recorded: fire intensity, forest type, fuel type, years since treatment, latitude, and longitude. We graphically explored the potential effects of these covariates to detect potential interactions between these variables and our results. The only covariate with explanatory power was longitude discretized as eastern vs western forests (east or west of Longitude 100 West). Twenty-six papers reported data from study sites in the western United States and six papers reported data from the eastern United States. We then ran all models using this discrete factor, east vs west as a moderator variable and dropped it from the model when it was not significant.

An important consideration in meta-analyses is the file drawer effect (e.g., Murtaugh, 2002). This is the bias due to the elevated rate of publication of statistically significant results and rejection of non-significant results. We sampled across a wide range of sources (international to regional journals, government publications), so the bias due to differential publication rates likely is minimal in the dataset we assembled. Additionally, we expect publication bias to be less important for this meta-analysis than for some others: for many of these studies, either fuel reduction or forest overstory restoration were the major focus of the work and the purpose of the applied treatments. Therefore, we argue that little effect on understory measures would not likely discourage publication. However, to assess prevalence of publication bias we visually examined funnel plots for each response variable and contrast (Sterne and Egger, 2001; Viechtbauer, 2010). We expect

Table 1
Papers with data included in this study. Some papers reported data from multiple sites (e.g., Waldrop et al., 2008) or from multiple treatments. These are distinguished by abbreviations in parentheses.

Paper	Location	Measurements	Thinning methodology	Fire intensity	Forest type	Years since treatment
Collins et al. (2007)	Blodgett Forest Research Station, California	Herbaceous Cover Shrub Cover Total Cover Total Richness Non-native Cover Non-native Richness	Thinning from below and rotary mastication	Medium	Mixed conifer	2
Dodson (2004)	Lubrecht Experimental Forest, Montana	Non-native Richness	NA	Low	Ponderosa pine	2.5
Dodson et al. (2007)	Lubrecht Experimental Forest, Montana	Total Richness Non-native Cover	Improvement/selection cutting and low thinning	Low	Ponderosa pine/ Douglas Fir	3.5
Fornwalt et al. (2010) (RL)	Pike National Forest, Colorado	Total Cover	NA	Low	Ponderosa pine/ Douglas fir	2
Fornwalt et al. (2010) (UH)	Pike National Forest, Colorado	Total Cover	NA	High	Ponderosa pine/ Douglas fir	2
Fornwalt et al. (2010) (UL)	Pike National Forest, Colorado	Total Cover Non-native Cover Non-native Richness	NA	Low	Ponderosa pine/ Douglas fir	2
Fornwalt et al. (2010) (UM)	Pike National Forest, Colorado	Total Cover	NA	Medium	Ponderosa pine/ Douglas fir	2
Fornwalt and Kaufmann (2014) (R)	Pike National Forest, Colorado	Herbaceous Cover Herbaceous Richness	NA	Low	Ponderosa pine/ Douglas fir	2
Fornwalt and Kaufmann (2014) (U)	Pike National Forest, Colorado	Herbaceous Cover Herbaceous Richness Total Cover Total Richness	NA	Medium	Ponderosa pine/ Douglas fir	2
Fulé et al. (2005)	Kaibab National Forest, Arizona	Total Cover Total Richness	NA	Low	Pine-oak	4
Huffman et al. (2013)	Kaibab NF, Arizona	Herbaceous Cover Herbaceous Richness Shrub Cover Shrub Richness Total Cover Total Richness Non-native Richness	Thinning from below	Low	Pinyon - juniper	5
Kane et al. (2010)	Challenge Experimental Forest, California	Herbaceous Richness Shrub Richness Total Richness Non-native Richness	Mastication using a rotary drum zstyle masticating head with fixed teeth	Medium	Ponderosa pine	4
Kerns et al. (2006)	Malheur NF, Oregon	Herbaceous Cover Non-native Cover Non-native Richness	NA	Low	Ponderosa pine	5
Knapp et al. (2006)	Sequoia National Park	Herbaceous Cover Herbaceous Richness Shrub Cover Shrub Richness Total Cover Total Richness	NA	Medium	Mixed conifer	3
Laughlin et al. (2008)	Coconino National Forest, Arizona	Total Richness	Thinning overstory vegetation	NA	Ponderosa pine	3
Mason et al. (2009) (BA)	Lincoln National Forest, New Mexico	Herbaceous Cover Total Cover Total Richness	Non-commercial thin with slash scattered	NA	Mixed conifer	1
Mason et al. (2009) (CO)	Lincoln National Forest, New Mexico	Herbaceous Cover Total Cover Total Richness	Non-commercial thin with slash scattered	NA	Mixed conifer	2
Mason et al. (2009) (SL)	Lincoln National Forest, New Mexico	Herbaceous Cover Total Cover Total Richness	Commercial harvesting with slash removed	NA	Mixed conifer	2

(continued on next page)

Table 1 (*continued*)

Paper	Location	Measurements	Thinning methodology	Fire intensity	Forest type	Years since treatment
Metlen and Fiedler (2006)	Lubrecht Experimental Forest, Montana	Herbaceous Richness Herbaceous Cover Shrub Cover Shrub Richness Total Cover Total Richness Non-native Cover Non-native Richness	Improvement/selection cutting and low thinning	Low	Ponderosa pine/ Douglas fir	2
Nelson et al. (2008)	Colville, Okanogan, Wenatchee National Forests, Washington	Herbaceous Richness Herbaceous Cover Shrub Cover Shrub Richness Total Cover Total Richness Non-native Cover Non-native Richness	Mechanical removal of small-diameter trees	Low	Ponderosa pine	8
O'Connor et al. (2013)	High Desert Ecological Province, Oregon	Herbaceous Cover	Cut-and-leave	NA	Pinyon - juniper	2
Phillips et al. (2007) (OH)	Central Appalachian Plateau, Ohio	Herbaceous Richness Herbaceous Cover Total Richness	Commercial thinning from below	Medium	Mixed hardwood	2
Phillips et al. (2007) (SA)	Southern Appalachian Mountains, North Carolina	Herbaceous Richness Herbaceous Cover Total Richness	Chainsaw felling of small, suppressed trees and shrubs	Medium	Mixed hardwood	2
Phillips and Waldrop (2008)	Clemson Experimental Forest, South Carolina	Total Richness	Cutting of trees to a residual tree spacing of ~6 m	Medium	*Pinus taeda/ Pinus echinata*	2
Provencher and Thompson (2014)	Smith Valley, Utah	Herbaceous Cover Shrub Cover	Lop and scatter	NA	Pinyon - juniper woodlands	4
Waldrop et al. (2010)	Southern Appalachian Mountain, North Carolina	Shrub Cover	Chainsaw felling of small trees and shrubs	Medium	Mixed hardwood	3
Waldrop et al. (2008) (GR)	Green River, North Carolina	Herbaceous Cover Shrub Cover Total Cover	Chainsaw felling of trees >1.8 m tall and all mountain laurel and rhododendron stems	Medium	Mixed hardwood	2
Waldrop et al. (2008) (OH)	Ohio Hills, Ohio	Herbaceous Cover Shrub Cover Total Cover	Commercial thinning from below	Medium	Mixed hardwood	2
Weekley et al. (2013)	Lake Wales Ridge State Forest, Florida	Herbaceous Cover	Logging: harvesting of all merchantable pine	Medium	Longleaf pine	2
Wienk et al. (2004)	Badger Game Production Area, South Dakota	Total Richness	Cutting of trees to leave a basal area of ~12 m^2/ha, trees and slash removed	Low	Ponderosa pine	2
Wolk and Rocca (2009)	Heil Valley Ranch, Colorado	Total Richness Non-native Richness	Chainsaw felling and hand crew or all-terrain vehicle skidding	NA	Ponderosa pine	3
Youngblood et al. (2006)	Blue Mountains, Oregon	Total Richness	Cut-to-length harvesting using a single-grip harvester and forwarder to remove merchantable live and standing dead and down material	Low	Ponderosa pine/ Douglas fir	5
Zald et al. (2008)	Teakettle Experimental Forest, California	Herbaceous Cover	Removal of trees between 25 and 76 cm diameter at breast height, while retaining at least 40% canopy cover	Medium	Mixed conifer	1.5
Zhang et al. (2008a)	Blacks Mountain Experimental Forest, California	Shrub Cover Shrub Richness Total Richness	Mechanical thinning, not otherwise specified	NA	Ponderosa pine	5

high heterogeneity across studies because of different methods and different measurement scales. To quantify heterogeneity, we calculated I^2:

$$I^2 = 100\% \times (Q - df)/Q$$

where Q is Cochran's heterogeneity statistic (Cochran, 1954) and df is degrees of freedom tested for significant heterogeneity (Viechtbauer, 2010). For comparisons with heterogeneity test p-values less than 0.05 and I^2 greater than 70%, we removed the outlier studies. We defined outlier studies as those outside the pseudo-confidence region with bounds $\hat{\theta}$ ±1.96SE, where $\hat{\theta}$ is the estimated effect or outcome based on the fixed-effects model and SE is the standard error value from the y-axis. For cases where outliers existed, we confirmed that our conclusions regarding the

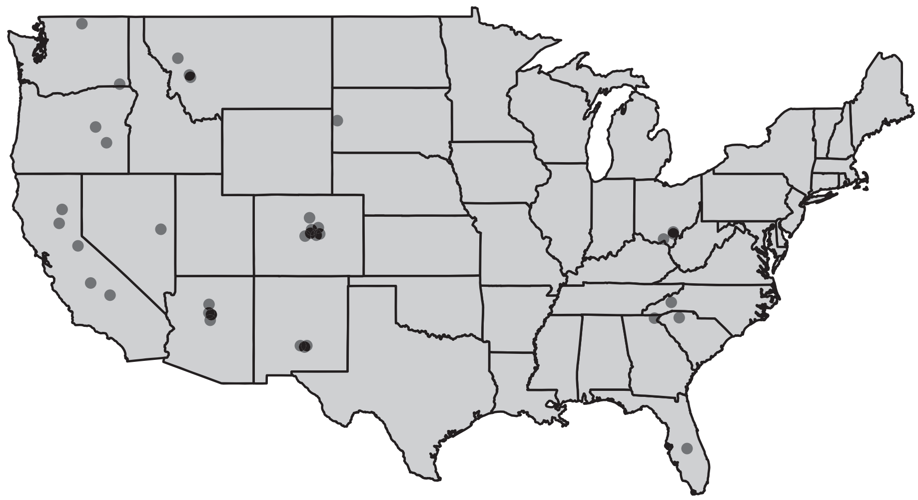

Fig. 1. Geographic distribution of data sources; see Table 1 for details of each study. Each circle represents a study site. We searched literature from North American forests that historically experienced frequent surface fire regimes (<20 years fire return interval). After vetting, only studies located in the USA remained.

Table 2
Contrast coefficient lower (lb) and upper (ub) confidence limits (90%) and adjusted p values for pairwise treatment contrasts (15 comparisons tested). Positive coefficients indicate response variable was higher in first treatment listed, negative indicates response was greater in second. We considered all hypotheses as one-tailed and therefore report 90% confidence intervals. We report experiment-wide adjusted p-values <0.1 as significant (in bold).

Response	Burn vs Control			Burn vs Thin			Thin vs Control		
	lb	ub	Adj. p	lb	ub	Adj. p	lb	ub	Adj. p
Non-native richness	0.240	1.05	>0.116	−0.880	−0.018	>0.1	0.534	1.29	**0.001**
Herb cover	−0.061	0.668	>0.1	−0.407	0.339	>0.1	−0.002	0.672	>0.1
Shrub cover	−1.36	−0.453	**0.014**	−1.56	−0.089	>0.1	−0.384	0.728	>0.1
Total cover	−0.624	−0.031	>0.1	−2.31	0.201	>0.1	−0.166	0.696	>0.1
Total richness	−0.327	0.850	>0.1	−0.338	1.36	>0.1	−0.062	0.977	>0.1

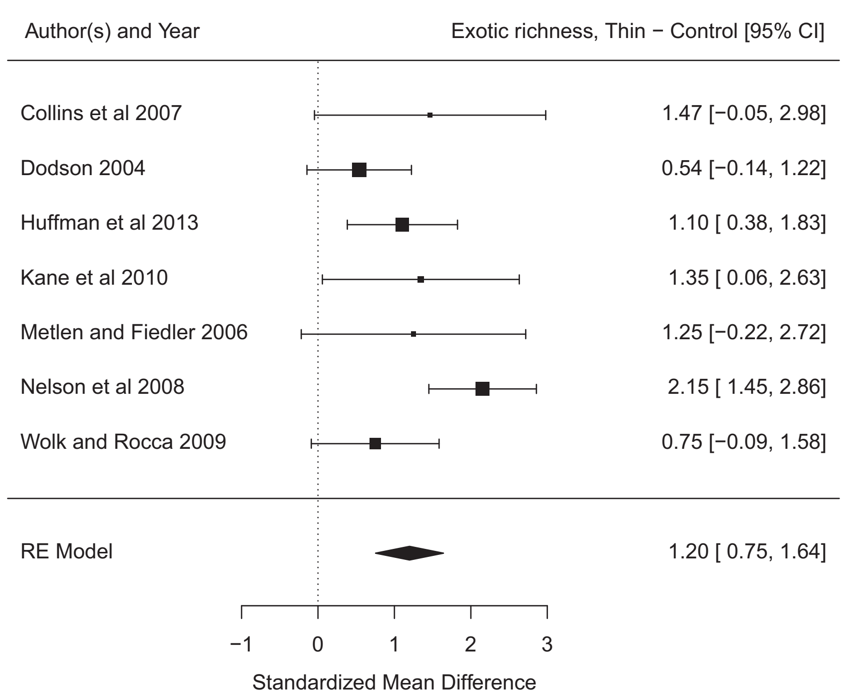

Fig. 2. Forest plot indicating standardized effect sizes (squares), standard deviations (error bars), and 95% confidence intervals of effect sizes for non-native species richness thinning vs control contrasts across relevant studies. The black polygon in the final row ("RE Model") indicates estimated average effect size with the horizontal extent of the polygon indicating the 95% $\alpha = 0.1$ confidence intervals of that estimate. Numbers on right of plot give these standardized effect sizes and upper and lower confidence intervals. Thinning treatments caused an increase in non-native species richness compared with control treatments (adjusted $p = 0.001$). Studies are listed on the left, see Table 1.

significant effect of treatments did not change after removal of the outlier, but all of our reported results include all available data even including such putative outliers.

3. Results

Our literature search and vetting resulted in 32 published studies. The length of time from treatment to sampling ranged from 1 to 8 years (mean = 2.8 years, median = 2 years). Only one paper reported data that were collected more than 5 years after treatment (Nelson et al., 2008). Although our search criteria included North America, after vetting, only studies located in the USA remained and our results were dominated by studies in the western US (26 western sites vs 6 eastern). Funnel plots for all comparisons are supplied in Appendix 1 (all studies). Removing outliers as described did not change results – the comparisons with outlier studies were comparisons in which the treatment effects were inconsistent. Total species richness showed the greatest heterogeneity across sites (Appendix 1).

3.1. Species richness

Non-native species richness was higher in thin treatments than in control treatments (Table 2, Fig 2, adjusted p = 0.001, N = 7), but this result is based entirely on western sites as no eastern thinning studies included this variable. Although non-native species richness was higher in burning treatments than in controls in all studies, this was not significant after Holmes correction (adjusted p = 0.116, N = 6).

Burning had a variable effect on total species richness (native and non-native) with increases relative to controls at eastern, but not western sites (no main effect, latitude significant modifier, adjusted p = 0.019, N = 12), but this result is based on 9 western and only 3 eastern sites (Fig. 3).

3.2. Percent cover

Burning decreased percent cover of shrubs compared to controls (Table 2, Fig. 4, adjusted p = 0.014, N = 7). There was no significant difference in shrub percent cover between thin and control treatments (N = 7), or between thin and burn treatments (N = 6). The effect of thinning on total understory cover was variable with little effect seen in most studies, but two eastern sites showed large negative effects of thinning on cover. This effect was significant after multiple comparison correction but was based on only these two eastern sites (Fig. 5, modifier p = 0.043, n = 9).

There was no significant difference in non-native percent cover or herbaceous percent cover among any of the treatment groups (Table 2). However, very few (4) papers reported percent cover data for non-native species (Table 1).

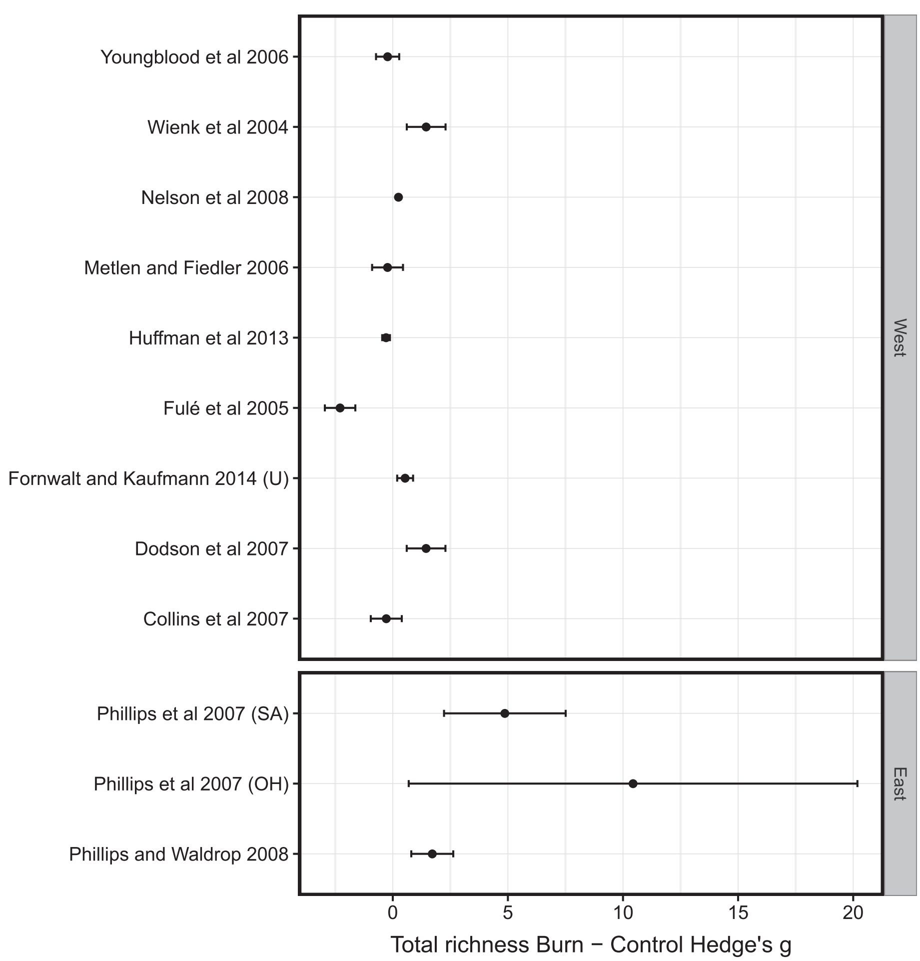

Fig. 3. Burn vs Control effect sizes for total species richness (Hedge's g, Gurevitch and Hedges, 2001). Mean effect sizes and 95% confidence intervals are shown for each study. Burning had a variable effect on total species richness with increases relative to controls at eastern, but not western sites (no main effect, latitude significant modifier, adjusted p = 0.019, N = 12).

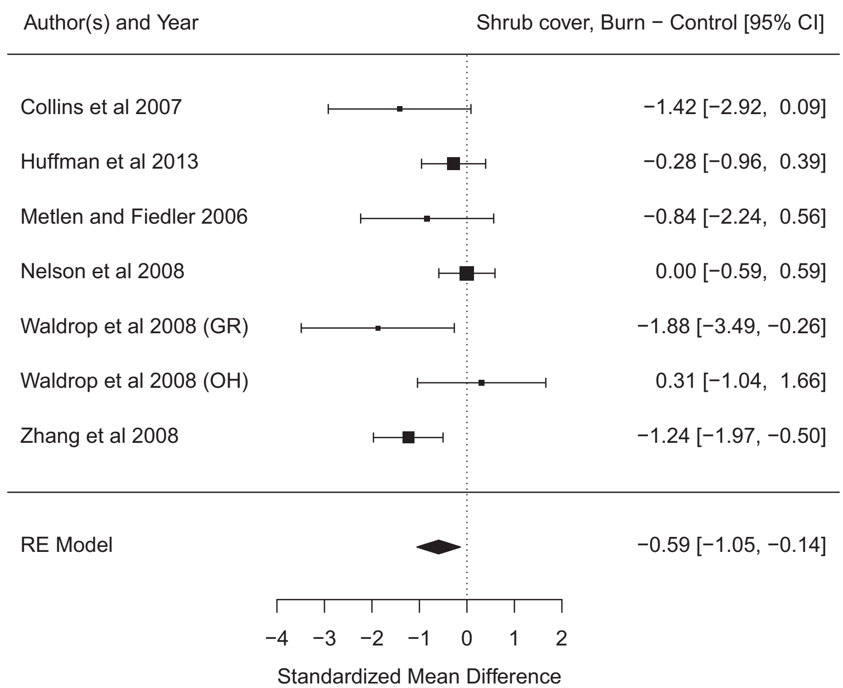

Fig. 4. Forest plot indicating standardized effect sizes (squares), standard deviations (error bars), and 95% confidence intervals of effect sizes for shrub cover burning vs control contrasts. Waldrop et al. (2008) report data from two sites and these were treated separately in our analyses (OH = Ohio site, GR = North Carolina site Table 1). The black polygon in the final row ("RE Model") indicates estimated average effect size with the horizontal extent of the polygon indicating the 95% $\alpha = 0.1$ confidence intervals of that estimate. Numbers on right of plot give these standardized effect sizes and upper and lower confidence intervals. Burning treatments caused a decrease in shrub cover when compared with control treatments (adjusted p = 0.014). Studies are listed on the left, see Table 1.

4. Discussion

Understory communities' response to fire and thinning was highly variable across the included studies and few comparisons had mean differences significantly different from zero. Our prediction that total species richness and herbaceous cover would increase in response to treatments was partially supported by the analysis (e.g., species richness increased after burns in eastern forests). These results combined with the Abella and Springer review (2015) suggest there is no consistent evidence that native herbaceous plants as a growth form respond to burning or thinning in the short-term. This does not mean that herbaceous understory plants do not respond to treatment, but it is likely there are species-specific responses that are lost when all taxa are lumped into the growth form category, herbaceous plants. We agree with Havill et al. (2015) and Keeley (2015) that future research needs to focus on developing a better mechanistic understanding of the way in which trait variation within plant functional types affects responses to disturbance. Pyke et al. (2010) provide one approach to predicting how fire will affect grassland plant populations that depends on the plant's life form and vital attributes for establishment and survival, and their interaction with fire regime. This approach could be expanded to predict responses of forest understory plants to both fire and thinning.

The most consistent effects of the treatments were the increase in non-native species following thinning and reduction in shrub cover following a burn. The significant reduction of the shrub layer during a burn which we predicted, was not observed in thinning, suggesting the two treatments may not be surrogates in the short-term. However, thinning treatments that include removal of shrub layer are not well represented in these studies so it is possible an overstory thin combined with mechanical shrub removal would mimic the effects of a burn on the shrub stratum. This requires further investigation.

There was also an interesting regional effect of treatments; at eastern sites, burning treatments increased total species richness relative to controls and thinning treatments decreased cover relative to controls. This second result is likely a consequence of regional differences in treatments: thinning treatments in the southeast often explicitly remove the shrub understory whereas, this is not a consistent feature of thinning treatments elsewhere (Schwilk et al., 2009). One possible explanation for the regional burn result is that the more mesic and productive eastern sites have faster growth dynamics and therefore, understory herbaceous species respond more quickly to a burn than at western sites. Western sites may exhibit similar responses in longer-term datasets currently not available in the literature.

Increase of non-native species due to disturbance is well established (Bartuszevige and Kennedy, 2009; Schwilk et al., 2009; Vilà and Ibáñez, 2011; Abella and Springer, 2015) and this analysis suggests thinning has a greater probability of creating conditions for non-native plants than burns in the short-term. The mechanism of this differential response is not clear and should be the subject of future investigation. Response of non-native plants to disturbance is likely a complex function of: (1) the plant's functional traits, (2) event-dependent factors such as seasonal timing or severity/intensity, (3) inter-disturbance intervals that affect length of recovery time between disturbances for sequestering resources or accumulating seed banks, and (4) environmental filters such as site climatic and edaphic conditions (Keeley et al., 2005; Havill et al., 2015; Keeley, 2015). There is insufficient information in the scientific literature to evaluate effects of disturbance intensity/severity, inter-disturbance intervals, site microclimate, and edaphic conditions on forest understories subjected to management treatments.

One concern of manipulating systems for conservation outcomes (e.g., fire surrogate treatments) is the risk of favoring non-native species. The increases in non-native species reported could

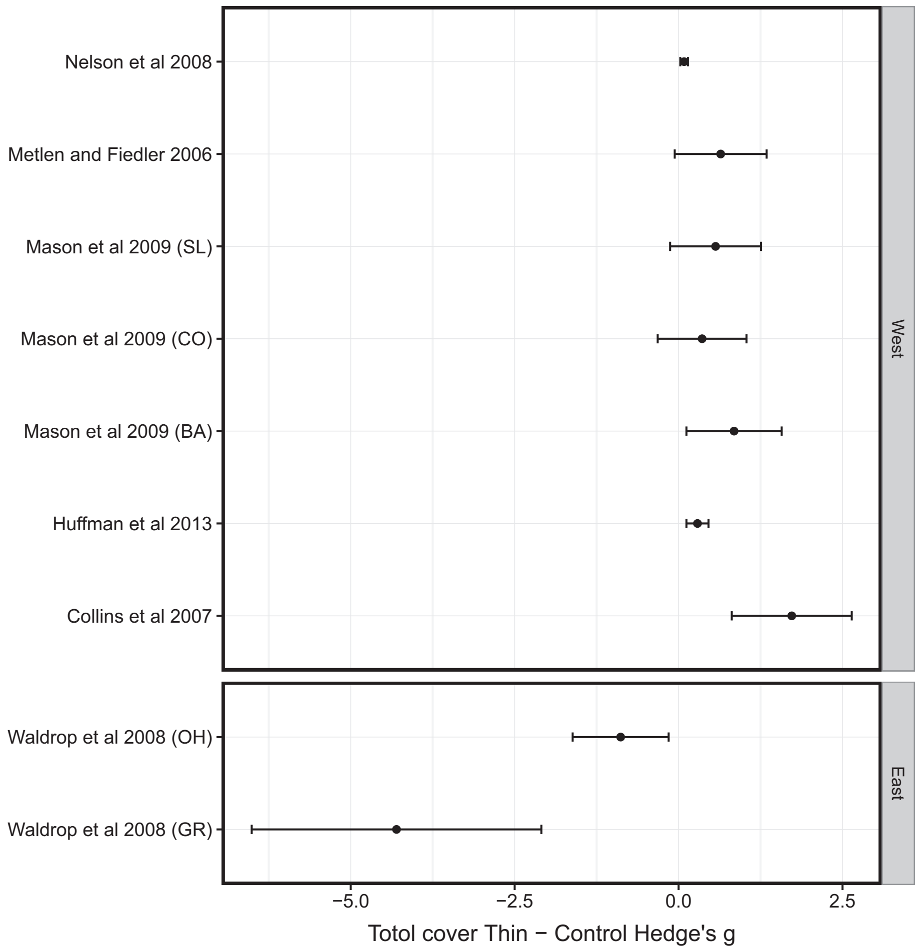

Fig. 5. Thin vs Control effect sizes for total understory cover (Hedge's g, Gurevitch and Hedges, 2001). Mean effect sizes and confidence intervals are shown for each study. Burning had a variable effect on total species richness with increases relative to controls at eastern, but not western sites (no main effect, latitude significant modifier, adjusted p = 0.002, N = 2015).

be biologically significant if allowed to increase in forest understory or if taxa were introduced by the treatment application. However, some forest stands may be at greater risk for establishment of non-native species. For example, non-native species are less likely to establish in the interior of forests, and more likely to establish in areas near other large invasions, fragmented landscapes, and human establishments (Bartuszevige et al., 2006; Vilà and Ibáñez, 2011; Johnson et al., 2016). Thus, an important consideration for treatment application is proximity to the wildland urban interface (WUI).

To prevent spread of non-native plants into forest ecosystems after treatment, managers should consider several management options including: no treatment, pre-treatment of non-native plants to reduce their abundance prior to treatment, seeding with native plants (Korb et al., 2004), reducing grazing by domestic livestock prior to and immediately after treatment (Keeley, 2006), or conducting a low impact disturbance (e.g. thinning alone, burning alone, or incremental treatments as opposed to both thinning and burning, or conducting the treatments all at once) (Dodson and Fiedler, 2006; Laughlin et al., 2008). Thinning treatments in particular can be modified to reduce soil disturbance (which facilitates invasion of non-native plants). Also, thinning in winter months when equipment will drive over snow will also minimize soil disturbance and thus, probability of invasion (Gundale et al., 2005).

The goal of returning forest understory composition to its range of historic variability may not always be feasible post-disturbance. Thus, to help guide the choice of management options we encourage managers to use current decision-making frameworks prior to designing an intervention to control non-native plants post-disturbance. For example, Hobbs et al. (2014) present a decision framework, derived from recent research on novel ecosystems (Hobbs et al., 2009; Hulvey et al., 2013) which helps to identify relative values of ecosystems in different conditions and the management options available in each case. As seen from a landscape perspective, this framework provides a comprehensive approach to decision-making and management, including much-needed prioritization of resource allocations. Numerous alternative decision support approaches are also available (reviewed in Perring et al., 2015).

In conclusion, burning and thinning effects on understory plant species richness and cover were either minor or highly variable. These disturbances, particularly thinning, increases non-native species which is likely causing any increases in reported understory richness. It is important to understand the potential threat of these species to native understory species in forested landscapes which is beyond the scope of this study. Also, managers need to recognize that burns can reduce shrub cover which can take longer to recover than herbaceous plants. If these shrubs provide important wildlife habitat or the shrubs are species of conservation concern, thinning might be a better alternative for fuel reduction or forest restoration. We conclude that prescribed fire and thinning treatments can be used successfully to restore understory species richness and cover, but they can create different conditions and these potentially different outcomes need to be considered in the planning of a fuels reduction treatment.

There is now a wide literature on the effects of thinning and prescribed fire. Given the diversity of forest types represented and the enormous variation in treatments across studies, we expect very high study-to- study variation in these ecological responses and, therefore, generalization requires a large number of studies (Verheyen et al., 2016). Yet inconsistent reporting of results hindered our ability to include studies in this analysis. Abella and Springer (2015) also point this out in their qualitative review. We excluded over a dozen papers that lacked any reporting of variance. Others were excluded because the data reported were summarized in a way that was incompatible with the majority of studies (e.g., only pre or only post treatment differences reported [and therefore, differences between pre and post could not be calculated]). There is growing attention to the value of shared data (Tenopir et al., 2011). Although shared publicly available data is an important goal, traditional meta-analysis is possible using only published summary statistics. For generalization in forest ecology, reporting treatment means and measures of variance is a minimum that editors and reviewers should insist upon.

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Appendix A. Supplementary material

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.foreco.2017.03.010.

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