1   **The Effects of Thinning and Burning on Understory Vegetation in North America: A Meta-**

2   **Analysis**

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11   **ABSTRACT**

12   Management in fire-prone ecosystems relies widely upon application of prescribed fire and/or fire-

13   surrogate (e.g., forest thinning) treatments to maintain biodiversity and ecosystem function. The

14   literature suggests fire and mechanical treatments proved more variable in their effects on understory

15   vegetation as compared to their effects on stand structure. The growing body of work comparing fire and

16   thinning effects on understory vegetation offers an opportunity to increase the generality of conclusions

17   through meta-analysis. We conducted a meta-analysis to determine if there were consistent responses of

18   understory vegetation to these treatments in North American forests that historically experienced

19   frequent surface fire regimes (< 20 years fire return interval, FRI). Means and standard errors were

20   extracted from 32 papers containing data on the response of four understory functional groups

21   (herbaceous, shrub, non-native, and total) to thinning and burning treatments to calculate effect sizes.

22   Lack of replication and inconsistent reporting of results hindered our ability to include many studies in

23   this analysis. For each response variable (species richness and percent cover), we compared three

24   treatment pairs: burn vs control, thin vs control and thin vs burn. We calculated standardized mean

25   differences (Hedges' g) for each pair and tested if this differed from zero using a random effects model

26   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.

Keywords: burning, disturbance, fire surrogates, forest understory, fuels reduction, meta-analysis, non-native understory vegetation, thinning

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 of thinning and prescribed fire (Schoennagel et al. 2004, Schwilk et al. 2009, Schoennagel and Nelson 2011, Stephens et al. 2013).

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53   Although prescribed fire is often the preferred fuel reduction practice, forest managers often face social

54   and economic constraints on burning particularly when human safety and property might be

55   compromised. Additionally, in forests thought to have departed significantly from historical fire return

56   intervals, there is concern that introducing fire may result in unnaturally high intensity fire that may be

57   difficult to manage or may have negative ecological effects (but see Bond et al. 2012 and Fontaine and

58   Kennedy 2012). Therefore, mechanical fuel reduction methods have increasingly been used to reduce

59   fuels or restore historic stand structure (Crow and Perera 2004). Uncertainty regarding the relative

60   ecological effects of prescribed fire versus mechanical treatments has led to increasing attention on these

61   so-called "fire surrogates" such as the National Fire and Fire Surrogate study (Schwilk et al. 2009,

62   McIver et al. 2013).

63   Early forest management emphasized recruiting trees for commercial harvest. However, in the last half

64   of the 20th century, forest management practices have shifted focus to include managing for ecosystem

65   services, including biodiversity. In most forests, the majority of plant biodiversity is in the understory

66   herbaceous layer. In addition to harboring high diversity, understory herbaceous communities have

67   profound effects on other ecosystem services such as forest nutrient cycling (reviewed by Gilliam 2007).

68   Most attention has been paid to the effects of fire and mechanical treatments on forest structure and fuels

69   (e.g., Moghaddas et al 2008, van Mantgem et al 2011, Kreye and Kobziar 2015, and reviewed in Fulé et

70   al 2012); the extent to which mechanical treatments or thinning approximate effects of prescribed fire on

71   forest understory vegetation is not as well understood. Results from the National Fire and Fire Surrogate

72   study demonstrated that fire and mechanical treatments proved more variable in their effects on

73   understory vegetation as compared to stand structure (Schwilk et al. 2009). This is not entirely

74   surprising: although both fire and thinning remove overstory trees and allow increased light to reach

75   understory plants, extent of canopy removal varies with thinning intensity and fire severity. According

76   to a recent review by Abella and Springer (2015) treatments must reduce tree canopy cover to <30-50%

77   to elicit appreciable responses from the forest understory.

78   In addition to variable treatment effects on forest cover, fire and thinning modify the abiotic

79   environment differently. Fire restructures microsites and soils that many plants depend on for

80   germination and growth (Bond and van Wilgen 1996, Gundale et al. 2005, 2006, DeLuca et al. 2006).

81 Thinning, on the other hand, removes or rearranges (as opposed to consumes) vegetation and may alter

82 nutrient dynamics (e.g., Boerner et al. 2006). Many mechanical thinning methods also result in soil

83 disturbance and compaction that fire does not cause (Schwilk et al. 2009). The differences exhibited

84 between fire and fire surrogate treatments may result in differences in responses between native and

85 nonnative species, and in the percent cover and species richness in the herbaceous and shrub layers

86 (Dodson 2004, Wienk et al. 2004, Metlen and Fiedler 2006, Collins et al. 2007, Nelson et al. 2008,

87 Zhang et al. 2008b, Fornwalt and Kaufmann 2014). The growing body of work comparing fire and

88 thinning effects on understory vegetation (recently reviewed by Abella & Springer 2015 for mixed

89 conifer forests in North America) offers an opportunity to increase the generality of conclusions through

90 meta-analysis.

91 Our objective was to conduct a meta-analysis of the literature that investigated effects of thin and burn

92 treatments on understory species in North American. Specifically, we were interested in the degree to

93 which thin treatments mimic prescribed burn treatments, and to what extent burning or thinning differ

94 from control treatments. We tested three pairwise comparisons: thinning treatments versus controls,

95 burning treatments versus controls, and thinning treatments versus burning treatments for percent cover

96 and species richness in total species, exotic species, herbaceous species, and shrub species.

97 We tested the following hypotheses: 1) total species richness and cover of herbaceous understory plants

98 will increase in thin and burn treatments compared to controls as an effect of increasing light availability

99 (Wienk et al. 2004, Metlen and Fiedler 2006, Fornwalt and Kaufmann 2014); 2) total cover of

100 understory shrubs will decrease in response to burning, but not to thinning in the short term because

101 burning consumes understory shrubs and these are slower to respond to increased light than are

102 herbaceous species (Nelson et al. 2008, Zhang et al. 2008b); and 3) non-native plant species are often

103 favored by disturbance and we expect both thinning and burning to increase non-native species richness

104 and cover relative to controls with the greatest increases in thinning treatments as a result of greater soil

105 disturbance in thinning relative to burning (Dodson 2004, Collins et al. 2007).

106 **2. Materials and Methods**

107 *2.1 Literature Search and Vetting*

108 In May 2014, we performed a search of the scientific literature investigating effects of prescribed fire

109 and thinning treatments on understory vegetation. We used multiple databases: ISI Web of Science

110 (http://www.webofknowledge.com) and AGRICOLA (http://agricola.nal.usda.gov/) both of which

111 searched literature published since 1970 and Forest Science (http://www.cabi.org/forestscience/) which

112 searched literature published since 1939. We also supplemented these searches with a Google Scholar

113 search (http://scholar.google.com/) which, despite limitations in coverage, includes gray literature

114 publications as well as proceedings. In addition to these search engines, we included additional

115 references gleaned from publications found in the literature search.

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

117 - Understory AND native*
118 - Percent Cover AND native*
119 - Fire AND Understory*
120 - Understory AND exotic*
121 - Percent Cover AND exotic*
122 - Fire AND Percent Cover*
123 - Understory AND forb*
124 - Percent Cover AND forb*
125 - Burn* AND Understory
126 - Understory AND graminoid*
127 - Percent Cover AND graminoid*
128 - Burn* AND Percent Cover
129 - Understory AND shrub*
130 - Percent Cover AND shrub*
131 - Thin* AND Understory
132 - Thin* AND Percent Cover

133

134   The literature search from the databases yielded approximately 3,500 references, which were then vetted

135   for appropriate material. Documents were eliminated that dealt with medical issues (i.e., new treatments

136   for burn victims), investigations of ecological processes related to fire but not relevant to the scope of

137   this document (e.g., nutrient cycling, insect infestation), or modeling studies with little empirical data.

138   Because North American studies comprise the bulk of the literature and our power to examine larger

139   geographic patterns would be very low, we restricted our analyses to North America. We were

140   specifically interested in studies that collected quantitative data on the response of understory plants to a

141   prescribed fire or thinning treatment. We further narrowed our search to papers that specifically

142   compared thinning (understory or overstory) and prescribed fire. We excluded papers that only reported

143   the combined treatment of thinning and burning but did not include thinning or burning as separate

144   treatments. This vetting process yielded 57 references.

145   Because statistical reporting was not uniform across references, we performed a second round of vetting

146   to exclude papers that could not be placed in a quantitative meta-analysis. Papers that lacked replicated

147   treatments or failed to include a measure of variation (standard error, standard deviation or variance)

148   were excluded. In addition, studies that reported data collected 10 or more years after the thinning or

149   burning event were excluded because there wasn't sufficient literature to quantitatively examine long-

150   term effects of the treatments on understory vegetation. When multiple papers reported on the same

151   study (determined by location and study dates), we used only one of the published papers. Varying

152   levels of prescribed burn severity were inherent due to differences in species composition, terrain,

153   weather, and season. There were insufficient studies to quantitatively evaluate effects of burn severity on

154   understory vegetation. Thus, for studies that reported data for multiple levels of burning severity, the

155   moderate level of burning was selected for inclusion in this analysis. Varying forms of thinning and

156   mastication were used in thinning treatments. These included chainsaw, dragging a chain between

157   tractors, hand-thinning, thin-and-pile, thin-and-scatter, thin-only, thin-and-chip, partial-cut, and clear-

158   cut. Data were insufficient to quantitatively evaluate effects of various forms of thinning on understory

159   vegetation. Thus, for each paper that reported several types of thinning, data were selected that were as

160   close to a thin-only treatment as possible. If multiple thinning-only methods were used, we pooled data

161   across methods when possible. In some cases, separate papers reported results from the same

162   experiment (e.g., Phillips et al 2007 and Waldrop et al 2008). In these cases, we did include both as

163   separate studies if they reported different response variables but we avoided including duplicated sites in

164   any particular contast.

165   *2.2 Data Extraction and Analysis*

166   We investigated effect of burning and thinning treatments on two response variables that describe effects

167   on understory vegetation: species richness and plant cover. Each of these was recorded separately for

168   each of two growth forms (herbaceous species and shrubs) and separately for exotic species and all

169   species combined. This resulted in four vegetative categories (herbaceous, shrub, non-native and total)

170   and two response types (cover and species richness) for a total of eight possible response variables,

171   although not all were available in each study. Although the shrub and herbaceous categories are

172   exclusive of one another, the other categories can overlap. For example, in several studies, non-native

173   species are a subset of herbaceous species. When plots at multiple spatial scales were included in a

174   study, we selected the scale closest to 100 m^2 as this was the most commonly reported scale across

175   studies.

176   For each selected article, we extracted treatment means, sample size, and either standard deviations or

177   standard errors of the mean for each response variable from the results text, tables or figures as required.

178   In some cases, papers reported differences pre-post study and we used those differences rather than raw

179   richness and cover values. We calculated pairwise treatment effect sizes for each response variable for

180   three different pairwise comparisons: Thin vs Burn, Thin vs Control, and Burn vs Control. For each

181   comparison, we calculated the bias-adjusted estimate of the standardized mean difference ["Hedge's g",

182   a bias corrected version of Cohen's d (Gurevitch and Hedges 2001)] with the following equation:

183   (1) $g = \frac{X^E - X^C}{S^{EC}} J$

184

185   where X^E is the mean value of the response variable in the "experimental treatment" (Burn or Thin

186   depending on contrast), X^C is the mean value of the response variables in the "experimental control"

187   (Control or Burn depending on the contrast), S^{EC} is the pooled standard deviation of both groups, and J

188   is a term that corrects for bias due to small sample size (Gurevitch and Hedges 2001). The effect size, *g*,

189   can be interpreted as the difference between the cover or species richness of plants in treatments relative

190   to controls, measured in units of standard deviations.

191   We conducted all analyses in R (R Development Core Team 2013) using the *metafor* package

192   (Viechtbauer 2010). We assumed that true effect sizes varied across sites (treatments differ in details,

193   forests differ in absolute cover and richness, and plot sizes could vary) and therefore, we used a random-

194   effects models (Gurevitch and Hedges 2001). We parameterized models using restricted maximum-

195   likelihood. To test whether mean effect sizes for a comparison differed significantly from zero, we

196   assumed a normal distribution of effect sizes and their confidence intervals (Viechtbauer 2010). Because

197   of the large number of response variables and pairwise treatment comparisons, we used a Holm p-value

198   adjustment (sequential Bonferroni correction) to control for multiple comparison error rate. Our *a priori*

199   alpha-level was 0.1, which balanced the likelihoods of Type 1 and Type 2 errors.

200   A handful of compromises and assumptions were necessary to reconcile each reference into a single,

201   comparable format. When papers reported mean species richness or percent cover values and standard

202   errors for multiple years per treatment, we selected the data closest to the average years since treatment

203   for the overall data set (3 years). Because native species represented over 99% of species richness and

204   percent cover for papers in which both parameters were reported, 'native' was substituted for 'total'

205   species in cases where papers reported only 'native' (Collins et al. 2007, Dodson et al. 2007, and

206   Huffman et al. 2013).

207   The final data used to conduct this analysis were reported from a wide array of locations, forest types,

208   and ecological management histories (Table 1, Fig. 1). To account for this variation in sampling

209   methodologies and time-lines the following covariates were recorded: fire intensity, forest type, fuel

210   type, years since treatment, latitude, and longitude. We graphically explored the potential effects of

211   these covariates to detect potential interactions between these variables and our results. The only

212   covariate with explanatory power was longitude discretized as eastern vs western forests (east or west of

213   Longitude 100 West). Twenty-eight papers reported data from study sites in the western United States

214   and seven papers reported data from the eastern United States. We then ran all models using this discrete

215   factor, east vs west as a moderator variable and dropped it from the model when it was not significant.

8

216 An important consideration in meta-analyses is the file drawer effect (e.g., Murtaugh 2002). This is the

217 bias due to the elevated rate of publication of statistically significant results and rejection of non-

218 significant results. We sampled across a wide range of sources (international to regional journals,

219 government publications), so the bias due to differential publication rates likely is minimal in the dataset

220 we assembled. Additionally, we expect publication bias to be less important for this meta-analysis than

221 for some others: for many of these studies, either fuel reduction or forest overstory restoration were the

222 major focus of the work and the purpose of the applied treatments. Therefore, we argue that little effect

223 on understory measures would not likely discourage publication. However, to assess prevalence of

224 publication bias we visually examined funnel plots for each response variable and contrast (Stern and

225 Egger 2001, Viechtbauer 2010). We expect high heterogeneity across studies because of different

226 methods and different measurement scales. To quantify heterogeneity, we calculated I^2:

227 $I^2 = 100\% \times (Q - df)/Q$

228 where Q is Cochran's heterogeneity statistic (Cochran 1954) and df is degrees of freedom tested for

229 significant heterogeneity (Viechtbauer 2010). For comparisons with heterogeneity test p-values less

230 than 0.05 and I^2 greater than 70%, we removed the outlier studies. We defined outlier studies as those

231 outside the pseudo-confidence region with bounds $\theta \pm 1.96 SE$, where θ is the estimated effect or outcome

232 based on the fixed-effects model and SE is the standard error value from the y-axis. For cases where

233 outliers existed, we confirmed that our conclusions regarding the significant effect of treatments did not

234 change after removal of the outlier, but all of our reported results include all available data even

235 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 (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 Bonferroni correction (adjusted $p = 0.116$, N=6).

Burning had a variable effect on total species richness (native and exotic) 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 (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).

260 There was no significant difference in non-native percent cover or herbaceous percent cover among any

261 of the treatment groups. However, very few (4) papers reported percent cover data for non-native

262 species (Table 1).

4. Discussion

264 Understory communities' response to fire and thinning was highly variable across the included studies

265 and few comparisons had mean differences significantly different from zero. Our prediction that total

266 species richness and herbaceous cover would increase in response to treatments was partially supported

267 by the analysis (e.g., species richness increased after burns in eastern forests). These results combined

268 with the Abella and Springer review (2015) suggest there is no consistent evidence that native

269 herbaceous plants as a growth form respond to burning or thinning in the short-term. This does not mean

270 that herbaceous understory plants do not respond to treatment, but it is likely there are species-specific

271 responses that are lost when all taxa are lumped into the growth form category, herbaceous plants. We

272 agree with Havill et al. (2015) and Keeley (2015) that future research needs to focus on developing a

273 better mechanistic understanding of the way in which trait variation within plant functional types affects

274 responses to disturbance. Pyke et al. (2010) provide one approach to predicting how fire will affect

275 grassland plant populations that depends on the plant's life form and vital attributes for establishment

276 and survival, and their interaction with fire regime. This approach could be expanded to predict

277 responses of forest understory plants to both fire and thinning.

278 The most consistent effect of the treatments was the increase in non-native species following thinning

279 and reduction in shrub cover following a burn. The significant reduction of the shrub layer during a burn

280 which we predicted, was not observed in thinning, suggesting the two treatments may not be surrogates

281 in the short-term. However, thinning treatments that include removal of shrub layer are not well

282 represented in these studies so it is possible an overstory thin combined with mechanical shrub removal

283 would mimic the effects of a burn on the shrub stratum. This requires further investigation.

284 There was also an interesting regional effect of treatments; at eastern sites, burning treatments increased

285 total species richness relative to controls and thinning treatments decreased cover relative to controls.

286 This second result is likely a consequence of regional differences in treatments: thinning treatments in

287 the southeast often explicitly remove the shrub understory whereas, this is not a consistent feature of

288 thinning treatments elsewhere (Schwilk et al 2009). One possible explanation for the regional burn

289 result is that the more mesic and productive eastern sites have faster growth dynamics and therefore,

290 understory herbaceous species respond more quickly to a burn than at western sites. Western sites may

291 exhibit similar responses in longer-term datasets currently not available in the literature.

292 Increase of non-native species due to disturbance is well established (Bartuszevige and Kennedy 2009,

293 Schwilk et al. 2009, Vilà and Ibáñez 2011, Abella and Springer 2015) and this analysis suggests

294 thinning has a greater probability of creating conditions for non-native plants than burns in the short-

295 term. The mechanism of this differential response is not clear and should be the subject of future

296 investigation. Response of non-native plants to disturbance is likely a complex function of: 1) the plant's

297 functional traits, 2) event-dependent factors such as seasonal timing or severity/intensity, 3) inter-

298 disturbance intervals that affect length of recovery time between disturbances for sequestering resources

299 or accumulating seed banks, and 4) environmental filters such as site climatic and edaphic conditions

300 (Keeley et al 2005; Havill et al. 2015; Keeley 2015). There is insufficient information in the scientific

301 literature to evaluate effects of disturbance intensity/severity, inter-disturbance intervals, site

302 microclimate, and edaphic conditions on forest understories subjected to management treatments.

303 One concern of manipulating systems for conservation outcomes (e.g., fire surrogate treatments) is the

304 risk of favoring non-native species. The increases in non-native species reported could be biologically

305 significant if allowed to increase in forest understory or if taxa were introduced by the treatment

306 application. However, some forest stands may be at greater risk for establishment of non-native species.

307 For example, non-native species are less likely to establish in the interior of forests, and more likely to

308 establish in areas near other large invasions, fragmented landscapes, and human establishments

309 (Bartuszevige et al. 2006, Vilà and Ibáñez 2011, Johnson et al. 2016). Thus, an important consideration

310 for treatment application is proximity to the wildland urban interface (WUI).

311 To prevent spread of non-native plants into forest ecosystems after treatment, managers should consider

312 several management options including: no treatment, pre-treatment of exotic plants to reduce their

313 abundance prior to treatment, seeding with native plants (Korb et al. 2004), reducing grazing by

314 domestic livestock prior to and immediately after treatment (Keeley 2006), or conducting a low impact

315   disturbance (e.g. thinning alone, burning alone, or incremental treatments as opposed to both thinning

316   and burning, or conducting the treatments all at once) (Dodson and Fiedler 2006, Laughlin et al. 2008).

317   Thinning treatments in particular can be modified to reduce soil disturbance (which facilitates invasion

318   of non-native plants). Also, thinning in winter months when equipment will drive over snow will also

319   minimize soil disturbance and thus, probability of invasion (Gundale et al. 2005).

320   The goal of returning forest understory composition to its range of historic variability may not always be

321   feasible post-disturbance. Thus, to help guide the choice of management options we encourage

322   managers to use current decision-making frameworks prior to designing an intervention to control non-

323   native plants post-disturbance. For example, Hobbs et al. (2014) present a decision framework, derived

324   from recent research on novel ecosystems (Hobbs et al. 2009; Hulvey et al. 2013) which helps to

325   identify relative values of ecosystems in different conditions and the management options available in

326   each case. As seen from a landscape perspective, this framework provides a comprehensive approach to

327   decision-making and management, including much-needed prioritization of resource allocations.

328   Numerous alternative decision support approaches are also available (reviewed in Perring et al. 2015).

329   In conclusion, burning and thinning effects on understory plant species richness and cover were either

330   minor or highly variable. These disturbances, particularly thinning, increases non-native species which

331   is likely causing any increases in reported understory richness. It is important to understand the potential

332   threat of these species to native understory species in forested landscapes which is beyond the scope of

333   this study. Also, managers need to recognize that burns can reduce shrub cover which can take longer to

334   recover than herbaceous plants. If these shrubs provide important wildlife habitat or the shrubs are

335   species of conservation concern, thinning might be a better alternative for fuel reduction or forest

336   restoration. We conclude that prescribed fire and thinning treatments can be used successfully to restore

337   understory species richness and cover, but they can create different conditions and these potentially

338   different outcomes need to be considered in the planning of a fuels reduction treatment.

339   There is now a wide literature on the effects of thinning and prescribed fire. Given the diversity of forest

340   types represented and the enormous variation in treatments across studies, we expect very high study-to-

341   study variation in these ecological responses and, therefore, generalization requires a large number of

342   studies (Verheyen et al. 2016). Yet inconsistent reporting of results hindered our ability to include

13

343 studies in this analysis. Abella and Springer (2015) also point this out in their qualitative review. We

344 excluded over a dozen papers that lacked any reporting of variance. Others were excluded because the

345 data reported were summarized in a way that was incompatible with the majority of studies (e.g., only

346 pre or only post treatment differences reported [and therefore, differences between pre and post could

347 not be calculated]). There is growing attention to the value of shared data (Tenopir et al. 2011).

348 Although shared publicly available data is an important goal, traditional meta-analysis is possible using

349 only published summary statistics. For generalization in forest ecology, reporting treatment means and

350 measures of variance is a minimum that editors and reviewers should insist upon.

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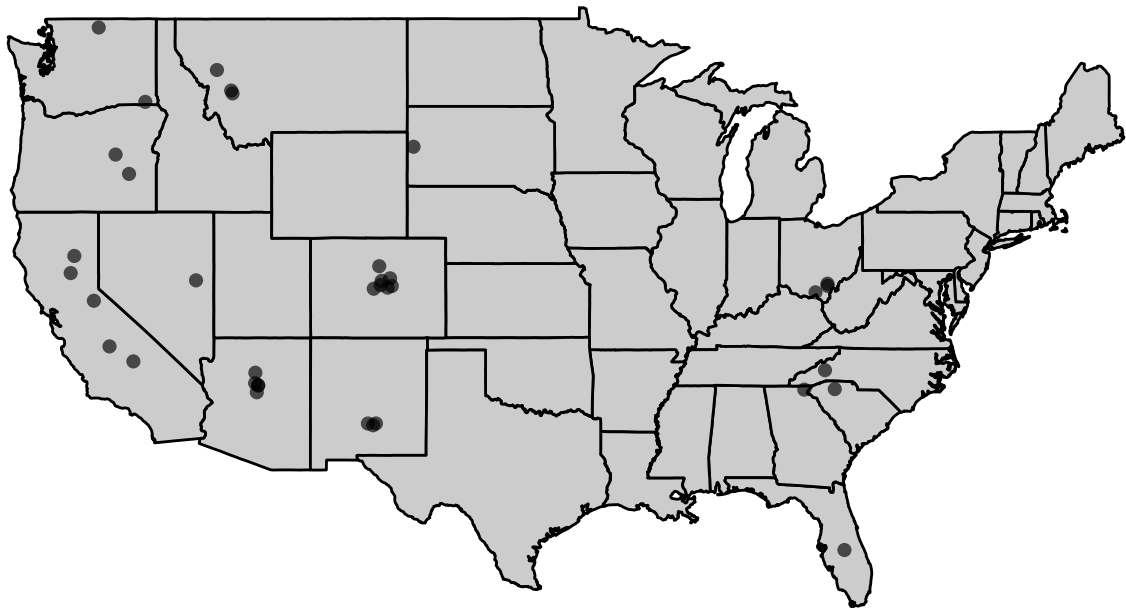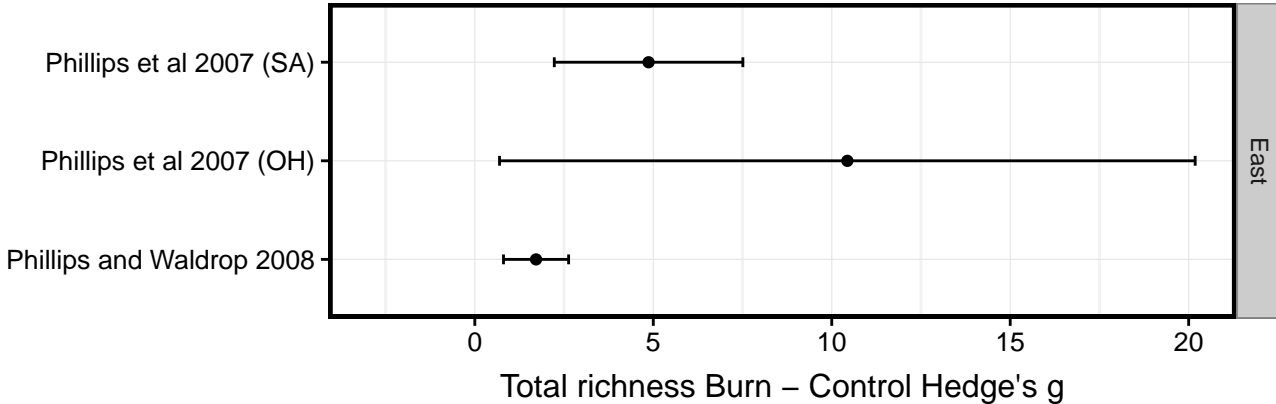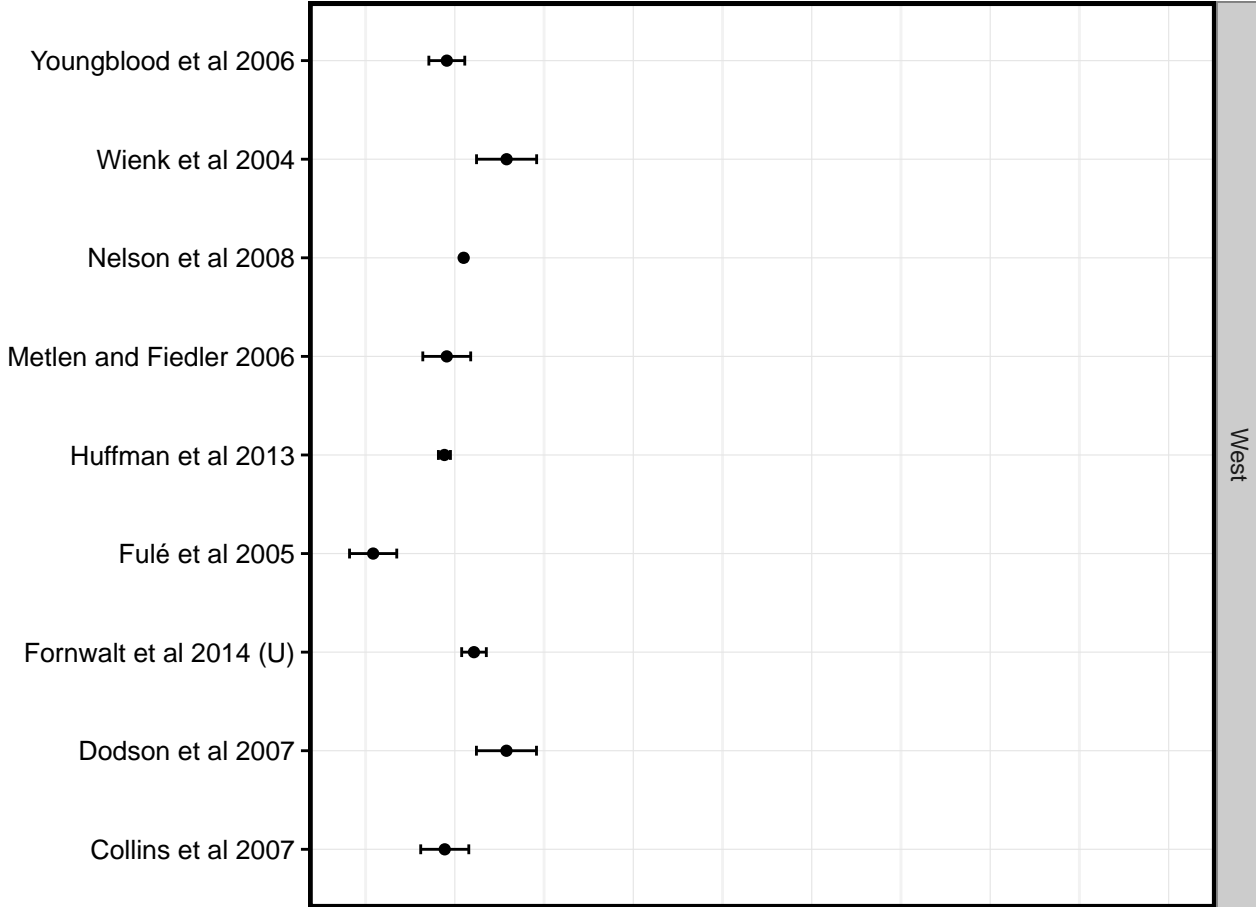525   Vilà, M. and I. Ibáñez. 2011. Plant invasions in the landscape. Landscape Ecology 26:461-472.

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Author(s) and Year	Exotic richness, Thin – Control [95% CI]
Collins et al 2007	1.47 [−0.05, 2.98]
Dodson 2004	0.54 [−0.14, 1.22]
Huffman et al 2013	1.10 [0.38, 1.83]
Kane et al 2010	1.35 [0.06, 2.63]
Metlen and Fiedler 2006	1.25 [−0.22, 2.72]
Nelson et al 2008	2.15 [1.45, 2.86]
Wolk and Rocca 2009	0.75 [−0.09, 1.58]
RE Model	1.20 [0.75, 1.64]

Standardized Mean Difference

Total richness Burn – Control Hedge's g
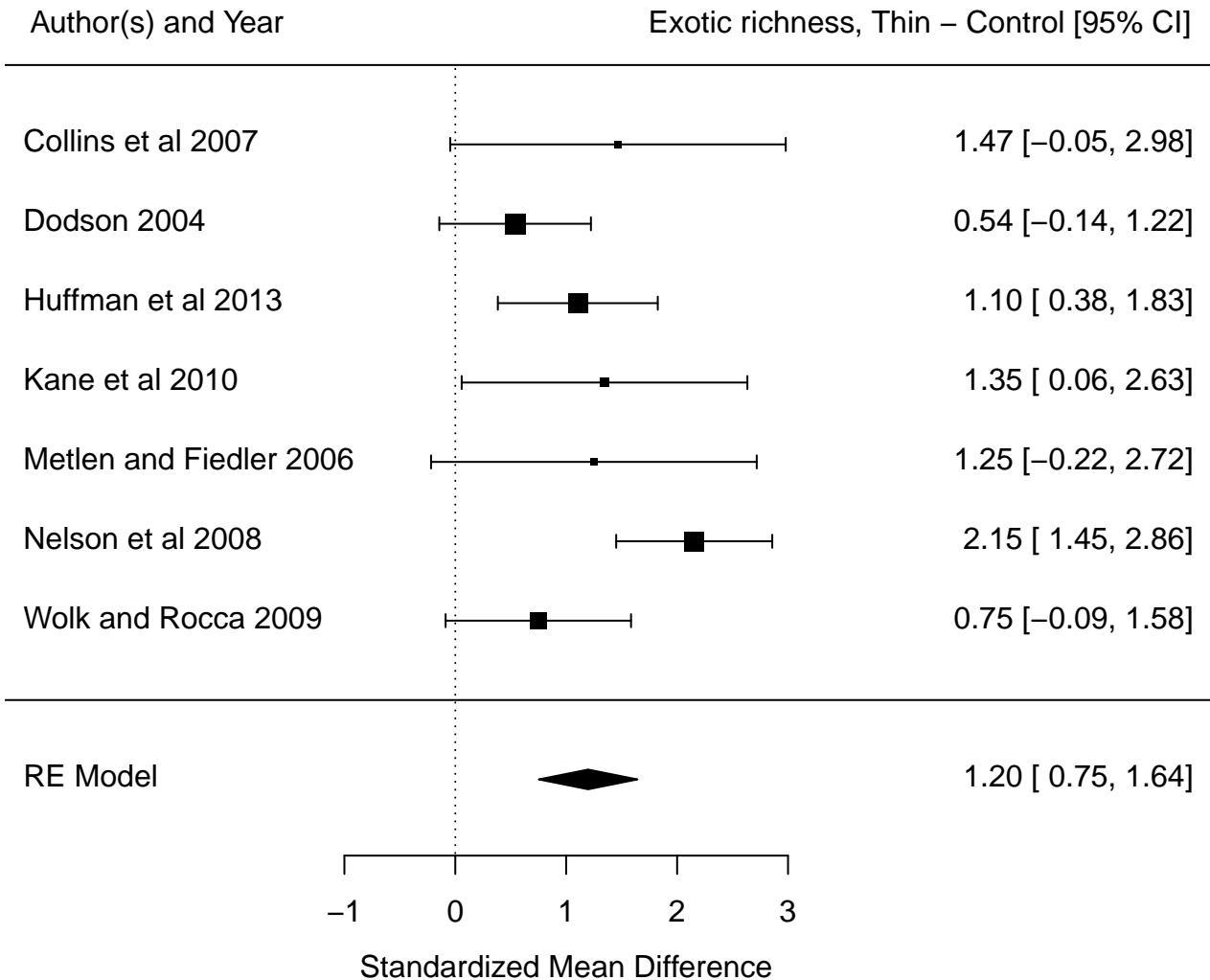
Author(s) and Year	Shrub cover, Burn – Control [95% CI]
Collins et al 2007	−1.42 [−2.92, 0.09]
Huffman et al 2013	−0.28 [−0.96, 0.39]
Metlen and Fiedler 2006	−0.84 [−2.24, 0.56]
Nelson et al 2008	0.00 [−0.59, 0.59]
Waldrop et al 2008 (GR)	−1.88 [−3.49, −0.26]
Waldrop et al 2008 (OH)	0.31 [−1.04, 1.66]
Zhang et al 2008	−1.24 [−1.97, −0.50]
RE Model	−0.59 [−1.05, −0.14]

Standardized Mean Difference

Total cover Thin − Control Hedge's g
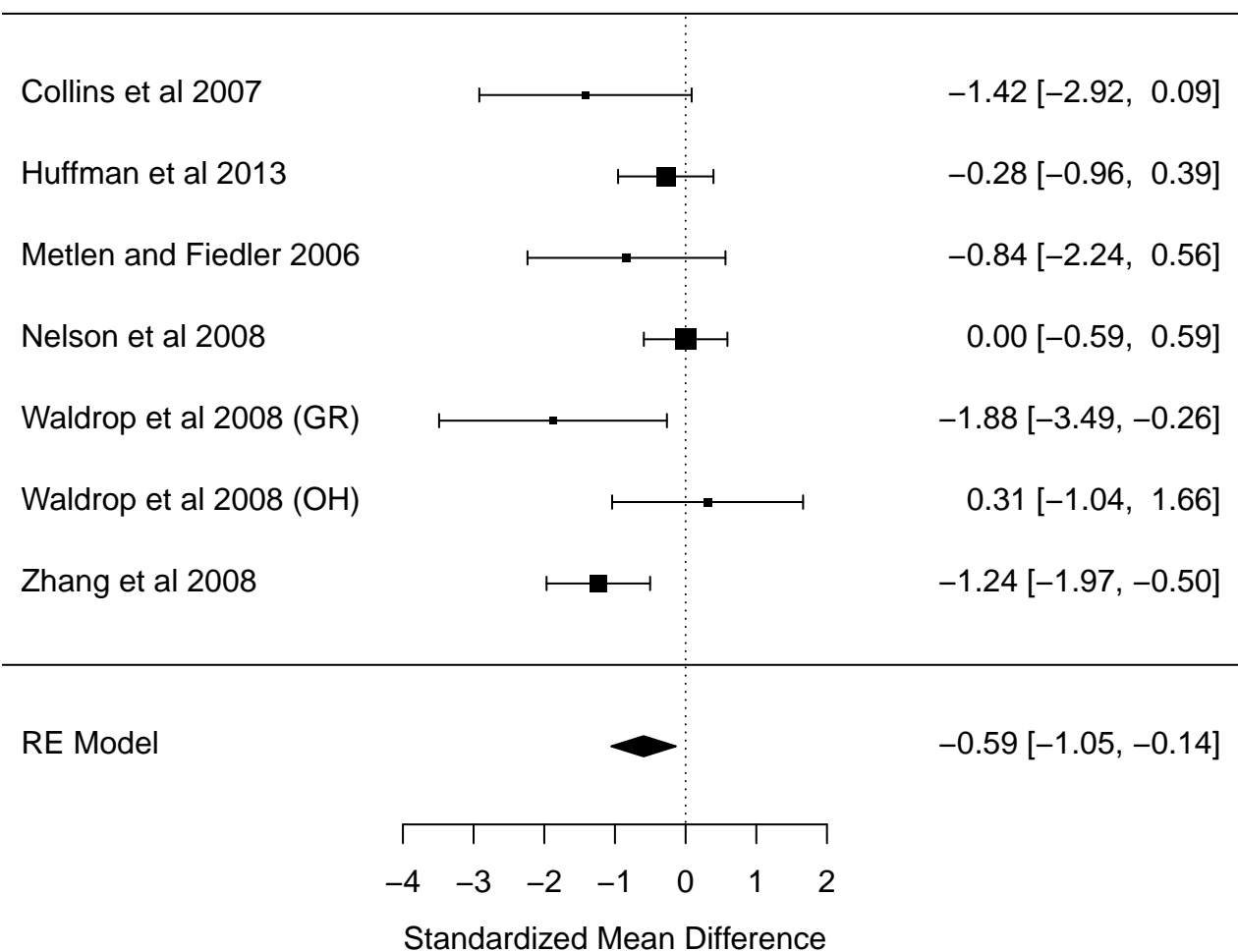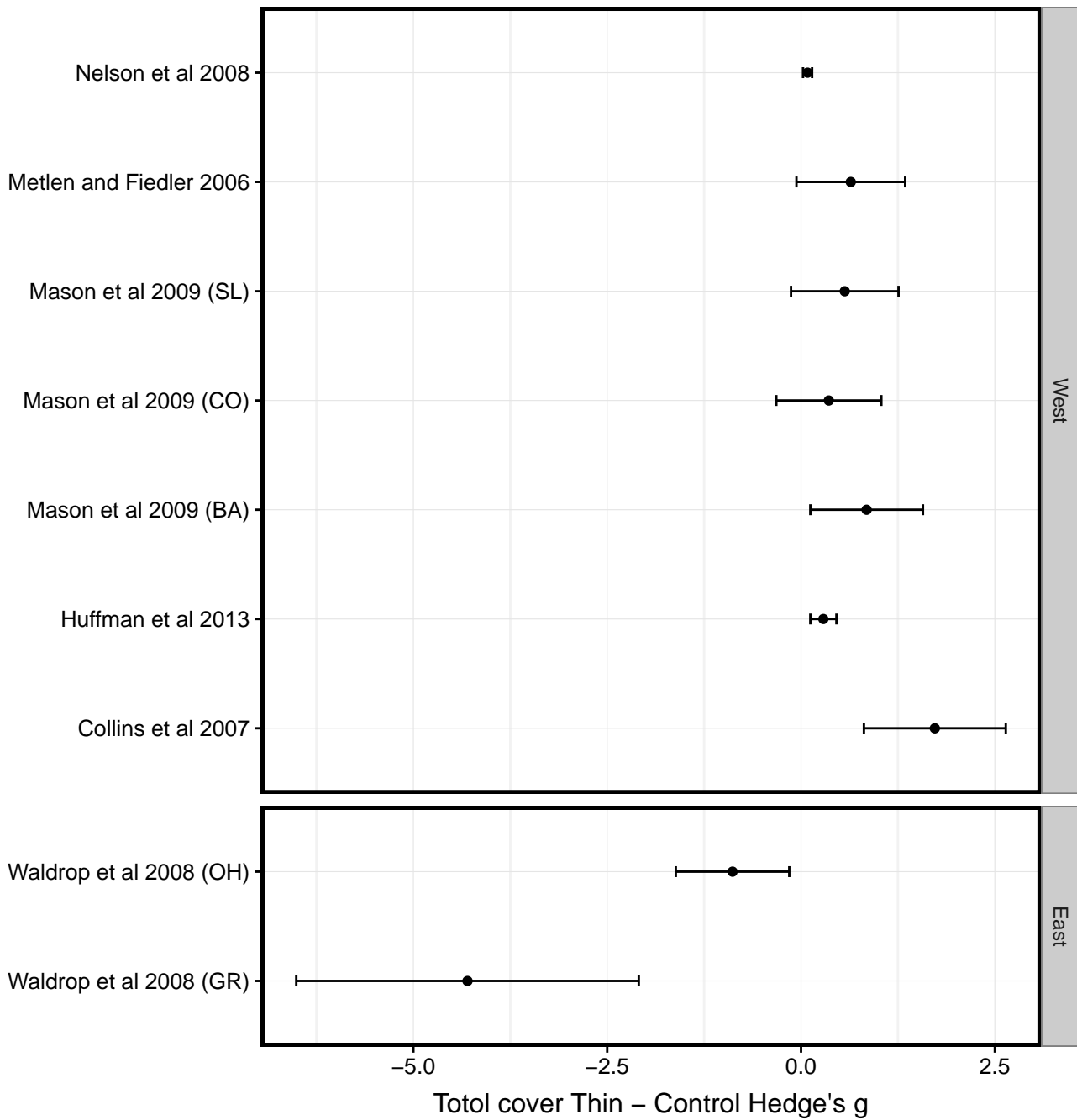
Tables

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 Exotic Cover Exotic Richness	Thinning from below and rotary mastication	Medium	Mixed conifer	2
Dodson et al 2007	Lubrecht Experimental Forest, Montana	Total Richness Exotic Cover	Improvement/selection cutting and low thinning	Low	Ponderosa pine/Douglas Fir	3.5
Dodson 2004	Lubrecht Experimental Forest, Montana	Exotic Richness	NA	Low	Ponderosa pine	2.5
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 Exotic Cover Exotic 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 et al 2014 (R)	Pike National Forest, Colorado	Herbaceous Cover Herbaceous Richness	NA	Low	Ponderosa pine/Douglas fir	2
Fornwalt et al 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	Thinning from below	Low	Pinyon - juniper	5

		Shrub Cover Shrub Richness Total Cover Total Richness Exotic Richness					
Kane et al 2010	Challenge Experimental Forest, California	Herbaceous Richness Shrub Richness Total Richness Exotic Richness	Mastication using a rotary drum style masticating head with fixed teeth	Medium	Ponderosa pine	4
Kerns et al 2006	Malheur NF, Oregon	Herbaceous Cover Exotic Cover Exotic 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
Metlen and Fiedler 2006	Lubrecht Experimental Forest, Montana	Herbaceous Richness Herbaceous Cover Shrub Cover Shrub Richness Total Cover Total Richness Exotic Cover Exotic 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 Exotic Cover Exotic 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 Exotic 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

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
exotic 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

Figure 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.

Figure 2: Forest plot indicating standardized effect sizes (squares), standard deviations (error bars), and 95% confidence intervals of effect sizes for exotic 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%α = 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 exotic species richness compared with control treatments (adjusted $p = 0.001$). Studies are listed on the left, see Table 1.

Figure 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).

Figure 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% α = 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.

Figure 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).