

Article (refereed) - postprint

Clark, Christopher M.; Richkus, Jennifer; Jones, Phillip W.; Phelan, Jennifer; Burns, Douglas A.; de Vries, Wim; Du, Enzai; Fenn, Mark E.; Jones, Laurence; Watmough, Shaun A. 2019. **A synthesis of ecosystem management strategies for forests in the face of chronic nitrogen deposition.**

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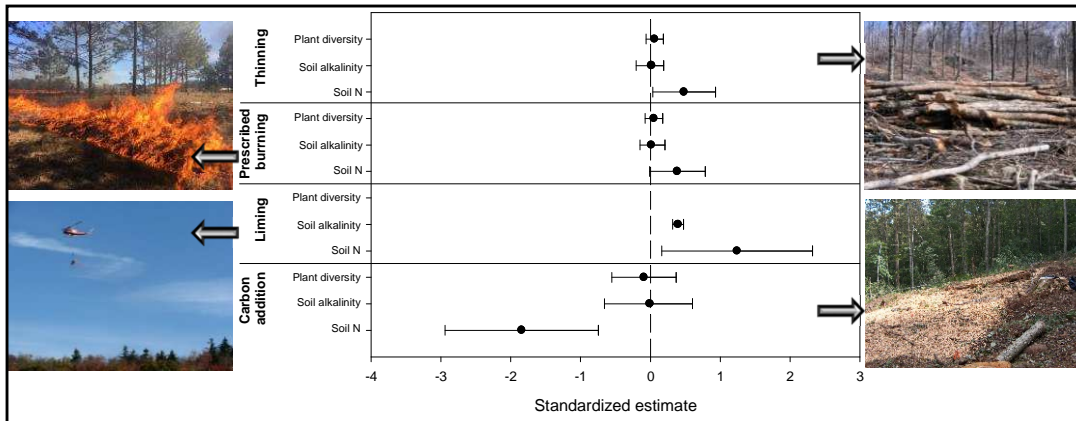
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Graphical abstract:



1 Title Page

2 Title: A synthesis of ecosystem management strategies for forests in the face of chronic nitrogen
3 deposition

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27

28 Abstract

29 Total nitrogen (N) deposition has declined in many parts of the U.S. and Europe since the 1990s. Even
30 so, it appears that decreased N deposition alone may be insufficient to induce recovery from the
31 impacts of decades of elevated deposition, suggesting that management interventions may be necessary
32 to promote recovery. Here we review the effectiveness of four remediation approaches (prescribed
33 burning, thinning, liming, carbon addition) on three indicators of recovery from N deposition (decreased
34 soil N availability, increased soil alkalinity, increased plant diversity), focusing on literature from the U.S.
35 We reviewed papers indexed in the Web of Science since 1996 using specific key words, extracted data
36 on the responses to treatment along with ancillary data, and conducted a meta-analysis using a three-
37 level variance model structure. We found 72 publications (and 2158 responses) that focused on one of
38 these remediation treatments in the context of N deposition, but only 31 publications (and 408
39 responses) reported results appropriate for our meta-analysis. We found that carbon addition was the
40 only treatment that decreased N availability (effect size: -1.6 to -1.8), while liming, thinning, and
41 prescribed burning all tended to increase N availability (effect sizes: +0.2 to +1.1). Only liming had a
42 significant positive effect on soil alkalinity (+10% to 80% across metrics). Only prescribed burning and
43 thinning affected plant diversity, but with opposing and often statistically marginal effects across
44 metrics (i.e., increased richness, decreased Shannon or Simpson diversity). Thus, it appears that no
45 single treatment is effective in promoting recovery from N deposition, and combinations of treatments
46 should be explored. These conclusions are based on the limited published data available, underscoring
47 the need for more studies in forested areas and more consistent reporting suitable for meta-analyses
48 across studies.

49 Capsule: Combinations of interventions in forests may be necessary to remediate soil N status,
50 soil acidity, and plant community diversity from the effects of long-term N deposition.

51 Keywords: nitrogen deposition, soil nitrogen availability, soil acidity, plant diversity, forest
52 management

53

54 Introduction

55 Enhanced atmospheric nitrogen (N) deposition remains one of the major stressors to
56 terrestrial ecosystems globally (Bobbink et al., 2010; Sala et al., 2000). In forests, N deposition
57 can lead to increased soil N availability (Aber et al., 1998), alterations in soil carbon and
58 biogeochemical cycling (Cheng et al., 2019; Tian et al., 2018), increased primary production (Du
59 and de Vries, 2018; Vitousek and Howarth, 1991), increases or decreases in tree growth and
60 mortality (De Vries et al., 2014; Du and de Vries, 2018; Horn et al., 2018; Thomas et al., 2010),
61 changes in tree nutrient status and vitality (De Vries et al., 2014), and increased N leaching to
62 downstream aquatic habitats (Driscoll et al., 2003a). Elevated N deposition, along with sulphur
63 (S) deposition, can also result in soil acidification, which can lead to nutrient imbalances, cation
64 and nitrate losses, reduced nutrient health of trees, and reduced rates of regeneration (Carter
65 et al., 2017; Driscoll et al., 2003b; Sullivan et al., 2013; Tian et al., 2018). Increased production
66 and aboveground biomass resulting from higher N availability can lead to decreased plant
67 diversity and/or shift species composition of understory community (Bobbink et al., 2010; Du,
68 2017; Hautier et al., 2009; Simkin et al., 2016; Verheyen et al., 2012). Long term N deposition
69 may even shift ecosystems away from typical N limitation (Vitousek and Howarth, 1991), and
70 toward limitation by other nutrients like phosphorus (P) (Goswami et al., 2018). The numerous
71 effects that N deposition can have on ecosystems, crossing environmental media (e.g., land, air

72 and water) as well as administrative jurisdictions, make the management of this environmental
73 stressor particularly challenging (EPA, 2010; Sutton et al., 2011).

74 There are many parts of the world where atmospheric N deposition is either unchanging
75 or continuing to increase, including the western U.S. (Houlton et al., 2013), industrializing parts
76 of Africa (Vet et al., 2014), and in most parts of south and eastern Asia (Duan et al., 2016;
77 Itahashi et al., 2018; Liu et al., 2011; Liu et al., 2013). Furthermore, emissions and deposition of
78 reduced N are not decreasing in many countries and may be increasing globally (Li et al., 2016;
79 Vet et al., 2014). On the other hand, deposition of total N and S have been declining in many
80 countries in Europe and in eastern North America after national and international air quality
81 policies were enacted to combat acid rain and ozone (Burns et al., 2011; Clark, 2018; Du et al.,
82 2014). Many of the air quality policies enacted to address these issues generally reduced S
83 more than N due to the focus on acidification. Thus, the role of N deposition (esp. reduced N) is
84 increasingly important now and into the future.

85 It is unclear whether decreasing N deposition alone is sufficient to promote forest
86 recovery (Gilliam et al., 2018; Schmitz et al., 2018; Stevens, 2016). Results from forests are
87 mixed, and demonstrate that some biogeochemical and physiological responses (e.g., nitrate
88 leaching, tissue N concentrations) may recover fairly quickly (Boxman et al., 1998b; Boxman et
89 al., 1995), while others (e.g., soil N, understory vegetation) may not (Boxman et al., 1998a; Van
90 Dobben and de Vries, 2017). Recovery from acidification appears to be beginning in some
91 forests in the eastern US (Lawrence et al., 2015) and Europe (Boxman et al., 2008; Johnson et
92 al., 2018; Verstraeten et al., 2012). However, not all forests or indicators show signs of recovery
93 (Gilliam et al., 2018; Lawrence et al., 2015; Verstraeten et al., 2017). Furthermore, recovery

94 may lead to other environmental issues such as the mobilization of soil carbon, N and possibly P
95 that may impact water quality downstream (Evans et al., 2012; Johnson et al., 2018; Monteith
96 et al., 2007; Oulehle et al., 2011; Oulehle et al., 2018; Stoddard et al., 2016). Regardless,
97 because of the increasing concern that ecosystems may not recover on their own, or may
98 recover but only over fairly long time scales (Johnson et al., 2018; Stevens, 2016), it is
99 increasingly apparent that management intervention may be required to assist forest recovery
100 from decades of elevated N deposition.

101 There are several remediation approaches that have been proposed or adopted to
102 reduce the effects of chronic N deposition and support ecosystem recovery. These include
103 prescribed burning, carbon addition, thinning, liming, litter or topsoil removal, mowing, grazing,
104 planting and harvesting of plant biomass, and replanting desired target species either as seed,
105 seedlings, or adults (Jones et al., 2016). Carbon addition involves the addition of a carbon
106 source that is either labile (e.g., sucrose) or more recalcitrant (e.g., sawdust) to induce
107 microbial immobilization of N and thereby reduce N availability to plants (Blumenthal et al.,
108 2003; Morgan, 1994; Torok, 2000). All of these approaches affect one or more processes that
109 influence recovery. Some approaches reduce soil N availability, either by enhancing soil
110 microbial immobilization (carbon addition, thinning with slash left onsite; Clark and Tilman,
111 2010; Eschen et al., 2006; Torok, 2000; Wolk and Rocca, 2009), or by direct removal of either
112 soil N (prescribed burning, litter/topsoil removal, planting and harvesting: Boerner et al., 2008;
113 Jones et al., 2016; Boxman and Roelofs, 2006), or of aboveground N (prescribed burning,
114 mowing, grazing, planting and harvesting, thinning with slash removal: Boerner et al., 2008;
115 Boxman and Roelofs, 2006; Jones et al., 2016). Other approaches reduce the acidity of the soil

116 whether from N or S (primarily liming: Błońska et al., 2015; Driscoll et al., 2001; Huang et al.,
117 2014; Lawrence et al., 1999; Lawrence et al., 2016; Likens et al., 1996). Other approaches make
118 habitats more suitable by either increasing light levels at the soil surface and opening
119 germination sites for colonization (prescribed burn, thinning, mowing, grazing: Ares et al., 2010;
120 Wolk and Rocca, 2009), or reintroduce propagules that may either be dormant in a deep seed
121 bank or locally extirpated (litter/topsoil removal, replanting: Bakker and Berendse, 1999; Jones
122 et al., 2016; Wolk and Rocca, 2009).

123 The effectiveness of remediation approaches for recovery from N deposition has
124 recently been assessed in European grasslands, heathlands, coastal habitats, bogs and fens
125 (Jones et al., 2016). While many approaches improved habitat suitability, most did little to slow
126 or reduce the accumulation of N in soils at current deposition rates. No parallel analysis has yet
127 been conducted for forests. We fill this key knowledge gap, and review the effects of four
128 remediation approaches in forests (i.e., prescribed burning, thinning, liming, and carbon
129 addition) on three indicators of recovery (i.e., reduction in soil N availability, increases in soil
130 alkalinity, and increases in understory plant diversity). Mowing, grazing, planting and
131 harvesting, and replanting of desired target species, are more common approaches in herb-
132 dominated communities (e.g., grasslands) than in forests, the focus of this study. We focus on
133 literature in the U.S. because of the intended focus on remediation from deposition in the U.S.,
134 but include seminal literature from Canada, Europe, and other areas to fill key gaps and provide
135 context.

136 Methods

137 Literature and data collection

138 We used a meta-analysis approach in this study (Borenstein et al., 2011; Gurevitch et al., 2001; Hedges
139 et al., 1999), because it facilitates the combination of results from multiple independent studies in order
140 to effectively increase sample size for a given research question. We searched literature databases
141 using Web of Knowledge, Google Scholar, and journal articles directly provided by co-authors (~60) and
142 from scientists at the U.S. Environmental Protection Agency (EPA), U.S. National Park Service (NPS), U.S.
143 Department of Agriculture Forest Service (USFS), and U.S. Geological Survey (USGS). Keywords for
144 searches included combinations of habitat (e.g., "Forest", "Deciduous forest", etc.), nitrogen and
145 nitrogen effects (e.g., "nitrogen," "eutrophication," "acidification"), and four management techniques of
146 interest (e.g., "liming," "prescribed burning," "carbon addition," "thinning," etc.) (Table S1). We
147 explored topsoil removal, mowing, and grazing, but these were not commonly found in forests and thus
148 were not included. We also included various synonyms of the core terms listed above (Table S1).

149 From the studies identified through literature searches, we selected those where remediation from N
150 deposition or improving N cycling was considered as part of the intention of treatment. Because of the
151 requirement of a focus on N, general studies that looked at these treatments for other reasons and that
152 did not discuss N in context were not included. We feel this is appropriate because we wanted to focus
153 on recovery from N deposition, rather than on the remediation approaches themselves, because
154 remediation approaches could be applied for reasons not pertinent to recovery from N deposition (e.g.,
155 clearing slash following forest dieback from drought, pests, or hurricanes; mine reclamation, etc.).
156 Results were further narrowed in Web of Science by publication years (1996 – present), countries (North
157 American and EU countries), and titles were reviewed to exclude papers that were not relevant. These
158 filtering steps reduced the number of publications from over 7000 to 72. Only a subset of these focused

159 on soil N availability, soil alkalinity, or plant community diversity (many addressed total biomass or other
160 end points), which further narrowed the literature set to 46.

161 Because we were using a meta-analytic approach, we required reporting of six parameters in tabular
162 form: (1) sample size of the treatment and control, (2) mean of the treatment and control, (3) standard
163 deviation/error of the treatment and control. This reduced the number of papers from 46 to 29, because
164 many papers did not publish all six required parameters, with the standard deviation/error lacking most
165 frequently. Some papers were excluded because they only published partial information in the main text
166 (e.g., means without standard deviations). We did not explore additional data mining software due to
167 time and resource constraints. Eleven of the 29 studies included were from the USFS Fire and Fire
168 Surrogates (FFS) program. These were treated as separate publications for each of the 11 locations
169 across the country where the FFS study design was implemented (McIver et al., 2012; McIver et al.,
170 2016).

171 Data processing

172 For the 29 studies used in the meta-analysis (Table S2), we identified records within the database with
173 control and treatment group mean, variance and sample size for three categories of response: soil N
174 availability, soil alkalinity, and plant community diversity. Each of these categories of response had
175 several metrics used in the literature to represent them. There were 10 for soil N availability
176 (ammonium, nitrate, nitrate N, inorganic N, net ammonification, N mineralization, net N mineralization,
177 nitrification, net nitrification, and mineralizable N), five for soil alkalinity (soil pH, base saturation, and
178 base cation exchangeable Ca, K, and Mg), and three for plant community diversity (species richness,
179 Simpson diversity index, Shannon diversity index) (Table 1). For the plant response category, non-native
180 species richness and diversity records were excluded where species origin information was available. We
181 used Simpson diversity where necessary to ensure increases in the metric were associated with

182 increases in diversity. For the soil N availability category, we excluded total soil N from the analysis for
183 several reasons: (1) we were interested in N availability rather than the total soil N pool, (2) total soil N
184 is a slowly changing variable that is likely not responsive over the study period for most studies (<5
185 years), and (3) including “slow variables” and “fast variables” in a meta-analysis can complicate the
186 portioning of variance and interpretation of results. For both the soil available N and alkalinity
187 categories, mineral and organic soil layers were included. All outcome measures were selected so that
188 responses were unidirectional within each system response (i.e., for soil N higher values indicated higher
189 soil N availability; for soil alkalinity higher values indicate greater alkalinity; for plant biodiversity higher
190 values indicate increased plant community diversity). Different studies reported multiple effect
191 categories, and/or multiple metrics within an effect category.

192 Overview of the literature surveyed

193 In total from the 29 publications included in the meta-analysis there were 408 effects reported
194 combining study, management method, and response metric for forests (Table 1, Table S2 and S3).
195 Most studies focused on the treatments of prescribed burning (17 studies and 152 effects) and thinning
196 (13 studies and 143 effects) (Table 2). Effects of these treatments were reasonably well distributed
197 across U.S. forests (Figure 1) and response categories owing to a common origin from the FFS (Table 2).
198 However, there was a notable absence of representative forests from Utah, Wyoming, and Idaho. Liming
199 was less intensively studied in the context of N deposition (9 studies and 101 effects), and almost
200 entirely focused on soil alkalinity (93 of 101 effects) and in the east (Figure 1). No studies reported the
201 effects of liming on plant diversity and only one on soil N availability. Most liming studies in the full
202 database of 72 references that examined the plant community assessed other responses such as plant
203 growth and tissue contents (e.g., N, Ca, P, etc.), rather than on biodiversity (Table S3). There were only
204 two carbon addition studies in forests (12 effects), and only one from the U.S. (Figure 1).

205

206 Statistical Analysis

207 Unique effect sizes were calculated for the 408 response records using a log transformed ratio of means
 208 (Eqn. 1) (Hedges et al., 1999).

$$209 \quad \ln RR = \ln\left(\frac{\bar{X}_T}{\bar{X}_C}\right) \quad (1)$$

210 and its variance as:

$$211 \quad V_{\ln RR} = S_{\text{pooled}}^2 \left(\frac{1}{n_T(\bar{X}_T)^2} + \frac{1}{n_C(\bar{X}_C)^2} \right) \quad (2)$$

212 with

$$213 \quad S_{\text{pooled}} = \sqrt{\frac{(n_T-1)s_N^2 + (n_C-1)s_C^2}{n_T+n_C-2}} \quad (3)$$

214 Where RR is the response ratio of treatment mean (\bar{x}_T) divided by the control mean (\bar{x}_C), S_{pooled} is the
 215 pooled standard deviation, n_T and n_C are the number of replicates for the treatment and the control, and
 216 s_N and s_C are the sample standard deviations for the treatment and the control, respectively. Log
 217 transformations are generally used in meta-analyses such as this because values generally follow a log-
 218 normal distribution (Borenstein et al., 2011).

219 Because selected outcome measures for soil N availability included zero and negative values (nitrate, net
 220 nitrification, and net N mineralization), the log transformed ratio of means is not applicable. Thus, soil N
 221 effect sizes were calculated using standardized mean difference (SMD), which is standard for responses
 222 that can be negative or positive (Eqn. 4).

$$223 \quad SMD = \bar{X}_T - \bar{X}_C \quad (4)$$

224 Thus, SMD was used for soil N availability, and log response ratios (lnRR) were used for the soil alkalinity
225 and plant responses. Effect sizes for SMD were measured in units of standard deviations, while for lnRR,
226 effect sizes were back transformed to percent changes. A small constant was added to metrics with
227 zeroes (i.e., soil nitrate).

228 A meta-analytic mixed-effects linear model was then fit to the data for each system response. All meta-
229 analytic models were specified using the “metafor” package (Viechtbauer, 2010) in the statistical
230 software program R (R Core Team, 2016). A three-level model structure was used to account for
231 different sources of variance in our database, including: (1) sampling variance, (2) among-study variance,
232 and (3) within-study variance for studies with multiple response metrics. In addition, moderator
233 variables representing all reported management practices were included in the model structure. This
234 modeling framework allows us to report estimates for each individual management action in the
235 dataset.

236 Meta-analyses were generally conducted at the effect category level (i.e., combining individual metrics
237 within the three response categories). Where there were sufficient data at finer levels of detail (e.g., for
238 individual metrics or subsets of metrics), additional separate meta-analyses were conducted. This
239 included two separate analyses within soil N availability (i.e., extractable N, net
240 nitrification/mineralization), two within soil alkalinity (i.e., soil pH, base cations [including base
241 saturation, exchangeable Ca, K, and Mg]), and two within plant diversity (i.e., species richness, diversity).
242 The binning of individual metrics into response subsets is shown in Table S4.

243 Results

244 Soil nitrogen availability response

245 Soil N availability (Figure 2, Table 3) was strongly reduced by carbon addition whether examining all
246 metrics ($P < 0.001$) or extractable N separately ($P = 0.002$). Prescribed burning tended to increase soil N
247 availability (all metrics, $P = 0.053$) through increased extractable N ($P = 0.024$), but had no effect on rates
248 of mineralization/nitrification. Liming and thinning had similar effects to one another, increasing soil N
249 availability (all metrics, $P = 0.021$ and $P = 0.031$, respectively), and tended to increase extractable N though
250 not at conventionally significant levels (i.e., $P = 0.073$ and $P = 0.137$ for liming and thinning, respectively,
251 Table S5).

252 Soil alkalinity response

253 As expected, soil alkalinity (Figure 3, Table 3) strongly and significantly increased with liming (all metrics,
254 +48.6%, $P < 0.001$). Individual response sets showed strong responses as well, with base cation
255 concentrations increasing by 82.2% ($P < 0.001$) and pH by 10.5% ($P < 0.001$). There were no other
256 treatment effects on combined or individual metrics for soil alkalinity (Table 3, Table S5).

257 Plant community response

258 Species richness tended to increase and diversity tended to decrease with either prescribed burning or
259 thinning (Figure 4, Table 3), though not at conventionally significant levels (Table 3). Carbon addition
260 had no effect on aggregated or individual diversity measures, and the variation of effects was wide
261 owing to the small number of studies in forests. Effects from liming on plant diversity were not
262 examined.

263 Discussion

264 Effectiveness of carbon addition to mitigate negative effects of nitrogen deposition

265 We found that carbon addition only improved one of the three indicators of recovery examined (i.e.,
266 decreased soil N availability). However, there were very few studies on the effects of carbon in forests
267 (N=5) and only two that reported results compatible for our meta-analysis. Cassidy et al. (2004) added
268 sawdust and sucrose (in addition to a separate lime treatment) to an upland forest in west-central
269 Massachusetts to test whether the invasiveness of Japanese barberry (*Berberis thunbergii*) was limited
270 by soil acidity or N availability. They found that carbon addition significantly reduced soil ammonium by
271 32-37%, and reduced N mineralization by 48-69%, though the latter result was not conventionally
272 significant ($P < 0.05$). Koorem et al. (2012) adjusted site fertility (i.e., increased with fertilizer addition
273 and decreased with sucrose addition) to an Estonian forest dominated by Norway spruce (*Picea abies*)
274 and common hazel (*Corylus avellana*). They found soil available N was relatively unchanged with sucrose
275 addition, though soil nitrate was not assessed and the plant community did show signs of decreased
276 shoots and species richness with sucrose addition. The three other studies not included in the meta-
277 analysis because of insufficient reporting had mixed results. Chapman et al. (2015) found that carbon
278 addition to a secondary oak forest in Pennsylvania led to reductions in nitrate and ammonium, but only
279 during certain times of the growing season, and Hunt et al. (1998) found little effect from carbon
280 addition to a mesic pine meadow site in Wyoming. This diversity of responses within years and across
281 precipitation gradients suggest covariation with climate as reported in grassland studies, with larger
282 reductions under drier rather than wetter conditions (Blumenthal, 2009).

283 Thus, although less studied in forests, there may be a growing body of literature suggesting that carbon
284 addition may have a remediation effect on reducing soil N availability (Blumenthal et al., 2003; Cassidy
285 et al., 2004; Clark and Tilman, 2010; Koorem et al., 2012; Morgan, 1994). The addition of a carbon-rich

286 source, whether labile (e.g., sucrose) or more recalcitrant (e.g., sawdust), induces the soil microbial
287 community to immobilize soil N, which reduces the available N in the soil to support plant growth
288 (Torok, 2000). There is much more literature from grasslands as opposed to forests on this effect. Thus,
289 even though the biogeochemical mechanisms are likely similar in forests, more work is needed to better
290 understand the efficacy of this approach for reducing soil N availability in forests, and how soil water
291 and climate affect the efficacy of treatment.

292 It is unknown whether carbon addition in forests has direct or indirect effects on soil alkalinity. There
293 may be direct effects through the alkalinity of the carbon source added, when added as wood chips
294 (contains base cations) as opposed to sucrose (i.e., no base cations). However, these are likely minor
295 relative to base cations in the mineral soil. Indeed, only Cassidy et al. (2004) and Chapman et al. (2015)
296 reported the effect of carbon addition on soil pH, and neither found a significant effect. There could be
297 indirect effects through the plant community through changes in plant growth and increases in
298 root:shoot biomass and root exudates to offset lower N availability (Eschen et al., 2006; Walker et al.,
299 2003), but these are likely minor.

300 In terms of the ultimate effects on the plant community, the results from the meta-analysis are
301 inconclusive because of small sample sizes (Table 1, Figure 4). However, closer inspection of the
302 literature suggest carbon addition may be a useful strategy. Koorem et al. (2012) found that the growth
303 of an understory forest specialist species (*Oxalis acetosella*) was enhanced with carbon addition,
304 provided the arbuscular mycorrhizal fungal community was not disturbed, while the growth of a
305 generalist species (*Prunella vulgaris*) was not significantly affected. Furthermore, Cassidy et al. (2004)
306 found that the growth of the forest invasive shrub Japanese barberry (*Berberis thunbergii*) was reduced
307 in the carbon addition treatment after accounting for initial biomass. These differential effects on target
308 species is often the goal of restoration approaches, where treatment is intended to reduce invasive
309 species and promote native species (Blumenthal et al., 2003). Both responses in Koorem et al. (2012)

310 and Cassidy et al. (2014) were associated with the effects of carbon addition on reducing soil available
311 N. These responses, however, were not large enough to significantly affect plant community diversity in
312 Koorem et al. (2012), and effects on diversity were not assessed in Cassidy et al. (2004).

313 These few studies for forests can be compared with a much larger number of studies on carbon addition
314 in grasslands (Blumenthal, 2009; Blumenthal et al., 2003; Clark and Tilman, 2010; Eschen et al., 2006;
315 Eschen et al., 2007; Paschke et al., 2000). Controlled greenhouse experiments show that carbon addition
316 reduces the growth of annual species and grass species more than perennial and leguminous species,
317 because the former group is more sensitive to soil N availability (Eschen et al., 2006). However, results
318 from field experiments are sometimes mixed. Reductions in total plant growth are consistently reported
319 in field experiments (Blumenthal, 2009; Blumenthal et al., 2003; Clark and Tilman, 2010; Eschen et al.,
320 2007; Paschke et al., 2000). However, species- and functional-group-specific results consistent with
321 predicted recovery (i.e., decreased cover of grasses, increased cover of forbs legumes, and increased
322 diversity) are not consistently found, and the restorative effect in some cases depends on climate, or an
323 initial disturbance to till in the carbon and/or stimulate the extant seed bank (Blumenthal et al., 2009;
324 Clark and Tilman, 2010; Eschen et al., 2007).

325 Carbon addition may promote the recovery of the desired plant community following N enrichment
326 through species- and functional-group-specific responses to reduced soil N availability, but there are
327 many uncertainties that remain. First, more studies are needed in forests, and for longer periods, across
328 a broad range of forest types and climates. Second, it is unclear whether the short-term effects
329 observed from carbon addition studies, whether as labile or more recalcitrant forms, are sustained over
330 long enough periods to induce permanent shifts in the plant community. Third, although it is
331 hypothesized that nutrient-demanding invasive species will be more hindered by carbon addition than
332 nutrient-efficient species, the results from field studies are mixed and may require additional treatments
333 in combination or an initial tillage to incorporate the carbon more fully into the soil. Fourth, because

334 there is no evidence of a direct or indirect effect of carbon addition improving soil alkalinity, a
335 combination of treatments may be required to restore soil alkalinity in addition to soil N and the plant
336 community. Finally, the efficacy of carbon addition is not only restricted to its ecological effectiveness,
337 but also cost and ease of application over a wide area. Lower cost carbon sources (e.g., sawdust or wood
338 chips over sucrose) may have the added benefit of longer lasting effects, though the magnitude of effect
339 may be lower than observed for more labile forms.

340 Effectiveness of liming to mitigate negative effects of nitrogen deposition

341 We found that liming had countervailing remediation effects, increasing soil alkalinity, but also
342 increasing soil N availability (all metrics). No study in the meta-analysis or the larger database examined
343 the effect of liming on plant diversity (Table 2). Those studies that did include plant end points focused
344 on other end points (e.g., tissue N, Ca, Mn, Mg; total biomass, etc., Table 3). Thus, most of the available
345 information focused on the effect of liming on soil alkalinity (and in the eastern U.S.), with other direct
346 or indirect effects on the ecosystem much less studied.

347 Liming has been well studied for decades as a treatment for mitigating the acidifying effects from N and
348 S deposition in the U.S. (Driscoll et al., 2001; Lawrence et al., 2016; Lawrence et al., 1999; Likens et al.,
349 1996), Europe (Błońska et al., 2015; Frank and Stuanes, 2003), and China (Huang et al., 2014; Li et al.,
350 2014). Given the historically high N and S deposition in the eastern U.S. and associated acidification
351 effects, the geographic emphasis in our study makes sense, though clearly areas in the west also deserve
352 attention. The application of lime restores nutrient and buffering base cations to the soil, enhances base
353 saturation on soil exchange sites, reduces the mobility of aluminum (Al) which is phytotoxic to plants,
354 and improves plant growth, health and recruitment. Our results of increased alkalinity are similar to a
355 much larger global meta-analysis focused on recovery from N and S acidification (Reid and Watmough,
356 2014). Because both N and S deposition can acidify soils (and N increasingly so as S deposition

357 decreases) lessons from Reid and Watmough (2014) and the associated literature are relevant. Reid and
358 Watmough (2014) found in their meta-analysis of 110 peer-reviewed publications, that six of the seven
359 end points evaluated had large significant and positive responses to liming and wood ash addition in
360 forests (i.e., soil pH, base saturation, tree foliar Ca, tree growth, ectomycorrhizal fungi root colonization,
361 and microbial indices), and only one did not have a significant response (soil C:N). They also found that
362 base cation addition had a larger effect on increasing soil pH in the organic soil layer than in the mineral
363 soil layer, consistent with the higher mineral content and slower responsiveness of the mineral layer to
364 surficial applied treatments. Reid and Watmough (2014) also found greater effects with larger treatment
365 magnitudes (e.g., with addition of lime over wood-ash, with the former having more base cations per
366 unit mass than the latter), in more acidic soils, and in the organic layer of younger stands. Thus, as with
367 other studies, liming is consistently shown to be an effective remediation treatment for soil acidification
368 whether from N or S deposition.

369 Soil N availability was not examined in Reid and Watmough (2014), and their finding of no significant
370 effect on soil C:N is not inconsistent with our results, as soil C:N changes much more slowly than soil
371 available N (Booth et al., 2005). Furthermore, our finding of an increase in soil N availability is consistent
372 with their reported increase in tree growth, even though increased tree growth could have been
373 induced by other factors as well (e.g., reduced Al mobility, increased base cation availability). Our results
374 on liming effects on soil N availability were only based on a single study (Cassidy et al., 2004), which
375 added lime, N, and sawdust-sugar to an upland forest in Massachusetts. They reported an increase in
376 nitrification and mineralization in the lime and N-addition plots relative to the sawdust-sugar, with the
377 controls in between. An increasing body of literature on recovery from acid deposition suggests that
378 increased N availability may be an unintended consequence of improvements in soil pH from reductions
379 in S deposition (Evans et al., 2012; Johnson et al., 2018; Monteith et al., 2007; Oulehle et al., 2011). In
380 more acidic soil, organic matter is more tightly retained in complexes with Al (de Wit et al., 1999), which

381 reduces solubility (Clark et al., 2006). Thus, with increases in pH induced either by liming or recovery
382 from acid deposition, more organic matter is available for mineralization that may then be measured in
383 the extractable pools (Evans et al., 2012). This has been observed in forested watersheds and in
384 downstream water bodies recovering from acidification, where organic losses from the forest floor leads
385 to N leaching to downstream waterways (Johnson et al., 2018; Monteith et al., 2007; Oulehle et al.,
386 2011).

387 It was somewhat surprising that no studies reported the effects of liming on plant biodiversity, given
388 that most forest diversity is in the understory herb layer (Gilliam et al., 2006; Gilliam, 2014). This could
389 reflect a historical bias for terrestrial liming studies to focus on acidification and soil or tree responses,
390 our filtering process that required a focus on N, or both. The intersection of these two sub disciplines
391 deserves more attention. Simkin et al. (2016) found that in more alkaline forested soils, N deposition
392 tended to monotonically increase understory species richness, and that in acidic forest soils N
393 deposition tended to increase then decrease understory species richness. Since liming is typically only
394 applied to acidified soils, liming may end up reducing plant diversity if the subsequent increase in N
395 availability induces competitive exclusion (Simkin et al., 2016). This may come at an additional loss of
396 soil available N to adjacent waterways during the process of recovery if the plant community cannot
397 take up the additional N. Increased N leaching can last a decade or longer (Oulehle et al., 2011), which
398 could impact downstream ecosystems until leaching levels decline to some new equilibrium. Oulehle et
399 al. (2011) found that decreases in S deposition in the Czech Republic between 1995 and 2009 (50 to 11
400 kg S ha⁻¹ yr⁻¹; no changes in N deposition), reduced N leaching from high levels (13 kg N ha⁻¹ yr⁻¹) to
401 almost no leaching (<0.2 kg N ha⁻¹ yr⁻¹) after 2006. Of this, net uptake by trees only accounted for 12.6%,
402 leaching losses for 23.9% and mineral soil N accumulation for 64.3% (Oulehle et al., 2011). A number of
403 studies from eastern North America also suggest a common pattern of declining nitrate leaching, as N
404 deposition decreases, but these patterns may vary due to legacy effects from historical acid deposition,

405 snow-pack dynamics, and other catchment-scale factors (Goodale et al., 2003; Judd et al., 2011;
406 Kothawala et al., 2011). Thus, elevated N leaching, though transient, may be a negative and unintended
407 consequence of liming forested soils.

408 Increases in soil N availability and leaching to waterways is not the indented outcome of remediation
409 efforts from exposure to N deposition. However, if acidification is the dominant effect, (whether from N,
410 S, or both), these effects may be inevitable without combining liming with other treatments. Only one
411 study in the meta-analysis (Cassidy et al., 2004) and two in the full database (plus Cleavitt et al., 2011)
412 had both carbon addition and liming as treatments, and neither of these examined them in
413 combination. Acidified areas often have suppressed plant growth (Carter et al., 2017; Schaberg et al.,
414 2002), and it may be difficult to synchronize the increased growth of desired plant species with
415 increased soil N availability that occurs once recovery from acidification begins. These desirable plant
416 species may not be present in the local community, and/or their growth may be sufficiently reduced to
417 preclude their ability to respond. However, combinations of liming and carbon addition with or without
418 seed addition may be a promising area for additional study. If the target plant community is still extant
419 and healthy, then perhaps liming alone might be sufficient to induce recovery. If not, then liming alone
420 may lead to either losses of N to waterways, or to sequestration of newly available N in undesirable
421 plant species, further inhibiting recovery. Either way, there may be a lag period between the recovery of
422 soil conditions and the ability of the plant community to respond, suggesting carbon addition as a
423 temporary measure to sequester N locally in the microbial pool. This sequestered N may then be taken
424 up by a recovering plant community. Thus, a careful diagnosis of the plant community, in addition to soil
425 nutrient and acidity status, is needed to determine a remediation plan for an affected area.

426 There are several uncertainties that remain regarding the efficacy of liming to mitigate the effects of N
427 deposition. First, more work is needed on endpoints other than soil alkalinity, including the processes
428 that control the rate and duration of increased soil N availability (Oulehle et al., 2011; Oulehle et al.,

429 2013), and whether these have positive or negative effects on plant community diversity. Second, a
430 deeper understanding is needed on how the type of base cation source, the magnitude of treatment,
431 and forest type and condition influence the soil alkalinity response (Reid and Watmough, 2014).

432 Effectiveness of prescribed burning to mitigate negative effects of nitrogen deposition
433 We found that prescribed burning had little utility in remediating the effects from N deposition, tending
434 to increase soil N availability over the short term, have no effect on long-term soil N availability or soil
435 alkalinity, and having countervailing effects on plant diversity. Prescribed burning is typically used to
436 reduce fuel loads in order to protect human settlements, minimize the spread of insects, disease, and
437 unwanted plant species, and promote the growth of desired plant species (Boerner et al., 2008; Ganzlin
438 et al., 2016; Stephens et al., 2009). In the context of recovery from N deposition, prescribed burning is
439 hypothesized to reduce the stocks of N in plant biomass and the upper soil horizon, reduce overall N
440 availability and promote the establishment of diverse assemblages of native species over invasive
441 species that may have come to dominate (Brockway, 1998; Cavender-Bares and Reich, 2012; Tilman,
442 1993). Prescribed burning also may increase light availability to the forest floor, create open sites for
443 new propagules, or trigger seed release and germination in fire-adapted species, thereby promoting the
444 establishment of new species that may have been locally extirpated (Tilman, 1993). On the other hand,
445 prescribed burning also directly removes plant cover, which reduces plant uptake of available N and can
446 lead to an increase in soil available N, N leaching, and/or erosion to nearby water bodies (Schoch and
447 Binkley, 1986). The effect of fire also depends on the frequency and intensity of events, with intense
448 events often damaging viable seeds and frequent events preventing their establishment. Overall, the
449 effect of prescribed burning on soil N availability likely depends on the duration of study. Over the short
450 term there are often observed net increases in soil N availability, from reduced plant uptake, increased
451 mineralization and nitrification, or N released from fuels (Ficken and Wright, 2017; Gundale et al., 2005;

452 Schoch and Binkley, 1986), which may transition to net decreases or no change in soil N availability as a
453 new equilibrium is established (Boerner et al., 2008; Ganzlin et al., 2016).

454 The short time duration of the FFS study (1-4 years), which made up the bulk of the prescribed burning
455 effects in our database (Table S2), can partly explain our reported increase in soil available N. The only
456 longer-duration study in the full database was Ganzlin et al. (2016) that re-measured soil N dynamics in
457 one of the FFS sites after 11 years (Lubrecht Experimental Forest in Montana, USA). Ganzlin et al. (2016)
458 found that the initial pulse in N availability after 1-4 years (Gundale et al., 2005), was not sustained after
459 11 years. Indeed, they found that all seven end points were not different from controls (O Horizon: NH_4^+ ,
460 NO_3^- ; Mineral soil: NH_4^+ , NO_3^- , net ammonification, net nitrification, net N mineralization). This is
461 consistent with the larger but shorter-duration meta-analysis of soil and vegetation N from the FFS study
462 (Boerner et al., 2008), which found that although total soil and vegetation N decreased with burning
463 after one year (due to direct removal by fire), these decreases relative to controls disappeared after
464 years 2-5. Additionally, other research suggests that fire removes only a small fraction of the large
465 mineral soil N reservoir and removal of soil N occurs only in the most severe wildfires (Johnson et al.,
466 2008). Only two other studies included in the meta-analysis also examined prescribed burning and soil N
467 availability. Ficken and Wright (2017) found from a study of three longleaf pine savanna sites in North
468 Carolina that the initial pulse in soil extractable N following a burn was short lived to only a few weeks.
469 Furthermore, only one of the three sites examined had sustained levels of higher soil extractable
470 ammonium, and that was in the site where the understory vegetation had not returned (Ficken and
471 Wright, 2017).

472 Prescribed burning was not found to affect any of the soil alkalinity. This lack of effect was reported
473 after 1-3 years (Boerner et al. 2008) and 11 years (Ganzlin et al. 2016). Thus, even though ash is typically
474 alkaline, prescribed burns are lower-intensity fires that do not volatilize minerals (Boerner et al., 2008;
475 Gundale et al., 2005) and thus do not lead to long term changes in soil alkalinity. Shorter-term studies

476 have found increases in alkalinity following burning (Stephens et al. 2009), but these appear to be short-
477 lived responses.

478 Prescribed burning did lead to subtle and countervailing changes in the plant community, with increases
479 in species richness ($P=0.054$) and decreases in diversity indices ($P=0.079$) (Table 3, Figure 4). These
480 results, though statistically weak, suggest that although the total numbers of species may increase, the
481 communities become more and more dominated by fewer species. These trends are contradictory to
482 the goals of remediating the effects of N deposition – i.e., increases in both native richness *and* diversity.
483 We hypothesize that the decrease in diversity measures is likely due to increased growth of extant
484 native species that were not removed by fire, and that were able to capitalize on increased soil available
485 N. Whether this is a favored outcome is dependent on the desirability of the native species. Either way,
486 given the short duration of the FFS study, (and that Ganzlin et al., 2016 did not assess the plant
487 community), it is unknown whether these changes persist over time. There was only one other study in
488 our dataset that assessed the effect of prescribed burning on plant diversity in forests (Brockway and
489 Lewis, 1997), though many assessed the effect of prescribed burning on other plant responses.
490 Brockway and Lewis (1997) found that although prescribed burning had little impact on overstory pines,
491 cover of grasses increased significantly by up to 2500% and Galberry shrub (*Ilex glabra*) declined 33%. In
492 a modeling study, Gimeno et al. (2009) concluded that only periodic prescribed burning, such as a burn
493 interval of every 15 years, is needed to remediate N-saturated forest soils. However, in some situations
494 burning too frequently can reduce ecosystem nutrient stocks resulting in decreased productivity
495 (Johnson et al., 2009; Raison et al., 2009).

496 Thus, it appears that prescribed burning is not an effective treatment for remediating soil N conditions
497 from chronic N addition, as: (1) short term losses from the fire itself or from the pulse in available N and
498 subsequent leaching merely redirects the eutrophication problem to other terrestrial or aquatic
499 ecosystems, (2) levels of total soil N are either not affected by fire, or are only affected short-term, and

500 (3) long term reductions in available soil N are not observed. Prescribed burning may be effective in
501 inducing plant community recovery, but this remains uncertain due to the short-term nature of most
502 studies included in our analysis. There are other goals that are clearly met with prescribed burning,
503 including reducing risk and severity of future wildfires (Fiedler et al., 2010; Stephens et al., 2009),
504 improved soil water status (Sala et al., 2005; Skov et al., 2004), and increased tree growth and
505 photosynthetic rates (Sala et al., 2005). Thus, prescribed burning is still a very useful management
506 practice for meeting other goals; however, remediation of long term N deposition may not generally be
507 one of them.

508 Prescribed burning was the most studied treatment in our assessment, with 17 studies and 152 effect
509 sizes across end points (Table 1). Nevertheless, significant uncertainties remain. Most of the information
510 was from the FFS study; and, although this is a very robust national study examining many factors across
511 forested areas in the U.S., it only ran for 4 years aside from individual site-specific follow up studies.
512 More work generally is needed on the effect of fire severity and frequency. Forest type also clearly
513 matters, as the ability of the target community to regenerate varies among forests.

514 Effectiveness of thinning to mitigate negative effects of nitrogen deposition

515 We found that thinning, like prescribed burning, had little utility in remediating the effects from N
516 deposition, tending to increase soil N availability over the short term, with no long-term effect on soil N
517 availability, and no or countervailing effects on the other two end points. Thinning is an approach that
518 along with prescribed burning, is utilized to reduce fuel loads to protect human settlements and other
519 factors not directly associated with N deposition (Boerner et al., 2008; Ganzlin et al., 2016). It has many
520 of the same ecological effects as prescribed burning (opening new sites, increasing light availability,
521 reduced plant N update), but with a few key differences. Thinning usually removes biomass in a patchy
522 manner, and this slash may or may not be removed from the site. Prescribed burns affect an area more

523 homogenously, and more of the biomass is volatilized compared with ash left on site. Thinning tends to
524 remove more aboveground biomass carbon and N while prescribed burning removes more total
525 biomass carbon and N because of the much larger removal of ground vegetation (Boerner et al., 2008).
526 These differences can lead to subtle differences in effects. If slash from thinning is left onsite, either as
527 fallen logs or chipped (Glitzenstein et al., 2016; Wolk and Rocca, 2009), then there is an abundant
528 carbon-rich material, which may lead to enhanced soil N immobilization by microbes relative to
529 prescribed burning.

530 We found that thinning had all the same directional effects as prescribed burning (Table 3, Figures 2-4).
531 Extractable N increased less and non-significantly for thinning (+0.4 units, $P=0.137$) compared with
532 prescribed burning that had a larger effect (+0.71, $P=0.053$). This likely occurred because burning
533 removed more total biomass and thus reduced plant uptake more than thinning (Boerner et al., 2008). A
534 weaker effect on extractable N from thinning compared with burning was probably not driven by
535 increased immobilization in thinned plots from carbon-rich slash left behind, as mineralization increased
536 though not significantly with thinning ($P=0.134$, Figure 2). The lack of a long-term effect on soil N cycling
537 from prescribed burning was also found for thinning (Ganzlin et al., 2016). Similarly, the lack of a long-
538 term effect from prescribed burning on forest floor and vegetation N was also found for thinning
539 (Boerner et al., 2008).

540 Thinning also increased plant richness and decreased diversity, though less than prescribed burning and
541 non-significantly (Table 3). The reason for these differences is unclear, but are likely driven by similar
542 processes as with burning, and the weaker effect is also likely driven by the weaker increase in soil
543 available N with thinning compared with burning. The only studies in our meta-analysis additional to the
544 FFS that focused on thinning effects on the plant community were Ares et al. (2010) and Wolk and Rocca
545 (2009). Wolk and Rocca treated a ponderosa pine site in Boulder CO, USA to thinning (with biomass
546 removal if feasible), and thinning with chipping where the wood chips were re-distributed onto the site.

547 They found that total understory cover increased only slightly with any treatment, and this was mostly
548 of non-native species. Treatment with- versus without-chippings responded differently in some
549 instances, with the chippings increasing the litter layer and reducing native and non-native plant growth.
550 Other studies mostly from grasslands have underscored the important influence that the litter layer can
551 have on the plant community, reducing germination and biodiversity because of lower light levels (Clark
552 and Tilman, 2010; Facelli and Pickett, 1991). In other ponderosa pine forests, thinning has been found to
553 increase the abundance and species richness of herbaceous understory vegetation (Laughlin et al., 2006;
554 Metlen and Fiedler, 2006), but not always (Metlen et al., 2004). On the other hand, Ares et al. (2010)
555 reported results from the western Oregon Cascades that were more consistent with the FFS and our
556 analysis, with increases in both native and introduced richness and cover, much more so from natives
557 than introduced species. Ares et al (2010) found that, comparing control with plots thinned to both high
558 density and medium density, early seral species increased by 3-15% in cover and 5-6 in richness, while
559 introduced species only increased by 0.6-0.8% in cover and 1.1-1.2 in richness. The difference between
560 these appears to be due to the degree to which the slash is removed (affecting the buildup of litter and
561 availability of germination sites), the initial cover of the plant community, and the availability of local
562 propagules of natives (affecting the ability to fill those new germination sites) (Ares et al., 2010; Wolk
563 and Rocca, 2009). Thinning seems to have little or no effect on understory vegetation in forests growing
564 at high latitudes or with understory plant communities having relatively low diversity (Ares et al., 2010).
565 Thinning is a form of disturbance, and weedy often non-native species are often early colonizers to a
566 disturbed site if they are locally present.

567 Thinning has also been applied and studied as an approach to minimize the large export of nitrate
568 observed to occur following clearcutting at sites with a history of elevated atmospheric N deposition
569 (Bormann et al., 1974; Burns and Murdoch, 2005). Studies have shown that thinning harvests can result
570 in either no change (Kastendick et al., 2012) or slight increases in export of nitrate (Clinton, 2011), but

571 where large increases have been measured, they have generally been much less than those that follow
572 clearcutting (Wang et al. 2006). Evidence suggests that there may be a harvesting intensity threshold
573 above which the relation between basal area removal and nitrate export steepens (Siemion et al., 2011;
574 Wang et al., 2006). Many factors such as the evenness of the harvest, tree species composition,
575 atmospheric N deposition levels, climate, soil drainage properties, and others affect the response of the
576 N cycle to thinning harvests (Gundersen et al., 2006). As found in prescribed burning studies, application
577 of a carbon source can maximize immobilization and minimize nitrate leaching following thinning
578 (Homyak et al., 2008).

579 Thus, overall thinning does not appear to be an effective approach for reducing the elevated N status of
580 forests from long term N enrichment for the same reasons as prescribed burning, but it may help in
581 some forests recover forest biodiversity depending on the type of forest and initial conditions of the site.
582 Although leaching was not specifically examined in this meta-analysis, forest thinning may be a
583 promising approach to reduce nitrate leaching over the long term as the forest regrows (though not
584 over the short term) as opposed to more intensive harvesting in areas of active forestry (Siemion et al.,
585 2011; Wang et al., 2006).

586 There are several uncertainties that remain with respect to the efficacy of thinning as a remediation
587 approach. Many of these are shared with prescribed burning discussed above (e.g., dependence on the
588 FFS, short time durations, frequency and severity of thinning, different types of forest types, soil, and
589 climate, etc.). However, some are unique to thinning and worth elaborating upon briefly. Removing
590 versus not removing slash was found to have significant effects on several aspects of community
591 recovery in Wolk and Rocca (2009). It appears that leaving slash on site may be better for soil health and
592 in some cases the plant community. However, the slash often is a merchantable product that creates
593 tradeoffs for decision makers in how to apply treatments. Either way, thinning is often more time
594 consuming and costly than prescribed burning, with costs as high as \$1000 per acre as opposed to as

595 little as \$86 per acre for prescribed burns. Thus, thinning is usually utilized when fire is not possible due
596 to hazardous burn conditions (Wolk and Rocca, 2009), and even then, its utility to remediate the effects
597 of N deposition may be limited. Nevertheless, as with prescribed burning, there are other management
598 goals that decision makers much weigh that thinning may meet when prescribed burning is not an
599 attractive option.

600 Conclusions

601 The relative global importance of N deposition as a stressor is likely to increase in the future, as N
602 deposition increases in Asia and Africa, and as N deposition declines less than S deposition in Europe,
603 the U.S., and Canada. Management interventions may be necessary to promote recovery, as these
604 ecosystems have been exposed to elevated N deposition for many decades (Clark et al., 2018), and
605 decreased deposition alone may not be sufficient to induce recovery over timescales desired by decision
606 makers (see Gilliam et al. 2019, Schmitz et al, 2019). Here we found that no single treatment was
607 effective in promoting recovery from N deposition for all three responses of interest (i.e., decreased soil
608 available N, increased soil alkalinity, increased plant biodiversity), suggesting combinations of
609 treatments should be explored, especially for situations where both acidification and eutrophication
610 impacts occur. In particular, the combination of carbon addition and liming may hold promise, with
611 carbon addition reducing N availability and liming increasing alkalinity. Our conclusions on the efficacy
612 of carbon addition in forests are largely inferential from the large body of work in grasslands and
613 deserves more attention in forests. Both liming and carbon addition may promote the recovery of the
614 plant community, but many uncertainties remain due to low numbers of studies of carbon addition in
615 forests overall, and low numbers of liming studies examining impacts on plant biodiversity.

616 Acknowledgements

617 We thank Robert Sabo and J Travis Smith for their review of an earlier version of the manuscript. This
618 research was supported by EPA's Office of Research and Development, Sustainable and Healthy
619 Communities National Program (SHC 3.3.1.3, SHC 4.61.4), and by RTI International (EPA contract EP-W-
620 13-010; Work Assignment 4-08; Amendment 2). All data from this study are available online through the
621 EPA Environmental Dataset Gateway (EDG, <https://edg.epa.gov>¹). The views expressed in this
622 manuscript are those of the authors and do not necessarily represent the views or policies of the U.S.
623 Environmental Protection Agency.

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¹ Permanent url generated following acceptance.

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937 Tables and Table Captions

938 Table 1: Response categories and individual metrics.

Response category	Individual metrics
Soil N availability	Ammonium, inorganic N, mineralizable N, N mineralization, net ammonification, nitrate, nitrate N, nitrification, net nitrification, net N mineralization
Soil alkalinity	pH, base saturation, exchangeable Ca, K, or Mg
Plant community diversity	species richness, Shannon diversity index, Simpson diversity index

939

940 Table 2: Summary of the literature included in the meta-analysis. Shown below are the number of
 941 studies out of 29 (and individual effects out of 408 in parentheses) that examined the effect of each
 942 treatment on each category of response (note one study could use multiple treatment levels, as well as
 943 address multiple response categories, and/or multiple metrics within a response category).

	Studies (effects)			
	Overall	Soil N availability	Soil alkalinity	Plant Diversity
Carbon addition	2 (12)	1 (8)	1 (2)	1 (2)
Liming	9 (101)	1 (8)	9 (93)	0 (0)
Prescribed burning	17 (152)	13 (46)	12 (40)	10 (66)
Thinning	13 (143)	10 (35)	9 (22)	10 (86)

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946 Table 3: Summary of statistical results from the meta-analysis. Shown below are the standard mean
 947 differences (SMDs) for soil N availability and back-transformed % difference (soil alkalinity and plant
 948 diversity for each of three responses for each of four treatments .

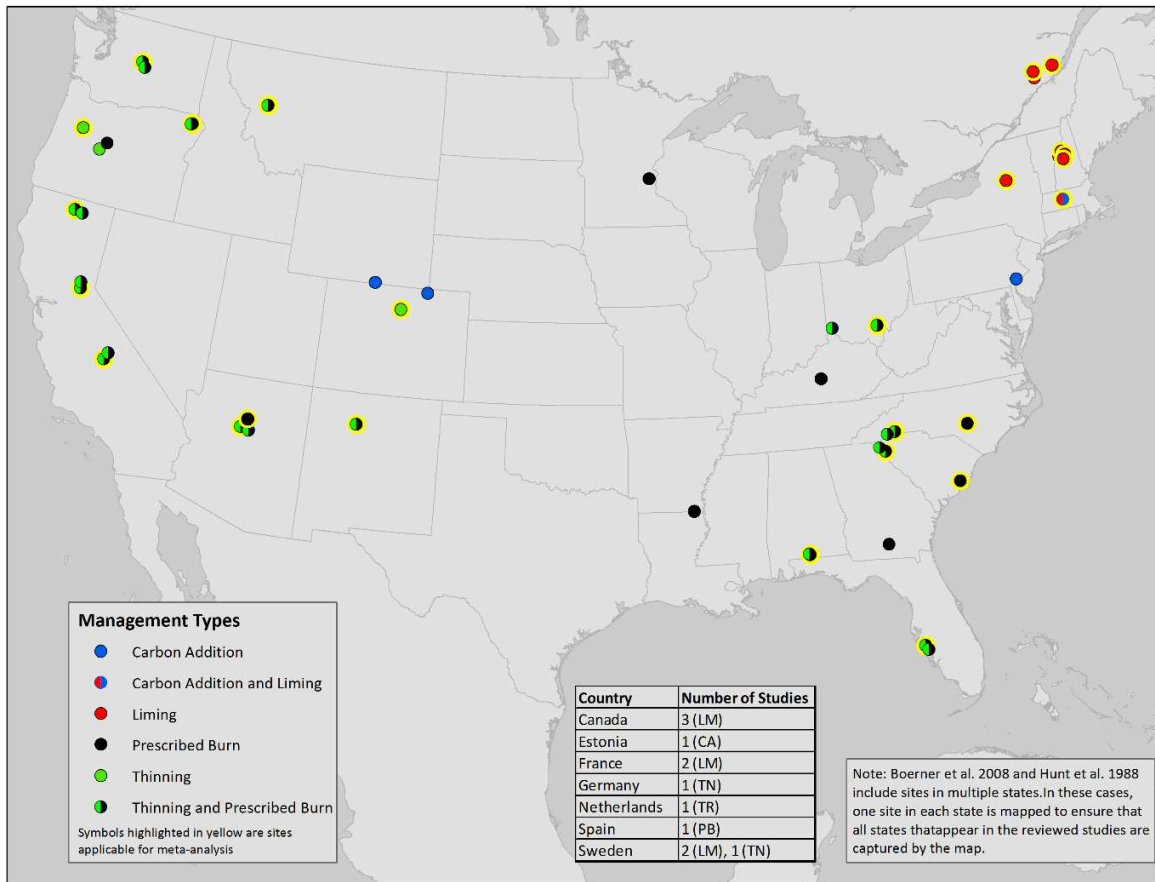
Response		Carbon		Prescribed	
category	Metric(s)	addition	Liming	burning	Thinning
Soil N availability (SMDs)	All	-1.8***	1.2*	0.4†	0.5*
	Extractable N	-1.8**	1.0†	0.7**	0.4
	Net Nitrification /Mineralization	NA	NA	0.1	0.4
Soil alkalinity (RR)	All	-2.7	48.6***	2.9	-0.9
	pH	-3.8	10.5***	2.8	-2.5
	Base cations	NA	82.2***	4.1	2.1
Plant Diversity (RR)	All	-8.9	NA	5.0	6.1
	Species richness	-6.6	NA	18.1†	12.6
	Diversity	-11.1	NA	-7.9†	-4.0

949 †, P<0.1; *, P<0.05; **, P<0.01; ***, P<0.001. NA indicates that this combination of treatment and effect
 950 was not present in the database.

951

952 Figures and Legends

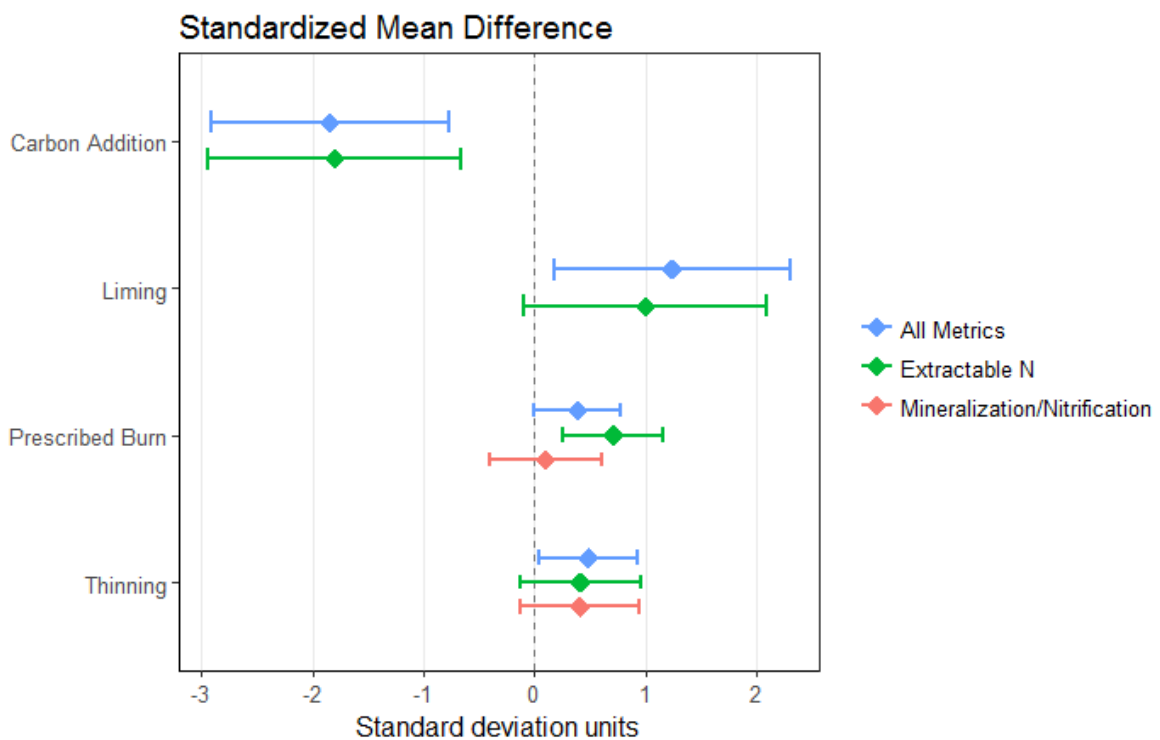
953 **Figure 1: Map of study locations and treatments.** Shown are the locations of the 46 studies included in
 954 the full database for the U.S. and Canada (other studies outside the map domain in the table inset), with
 955 the subset of 29 highlighted in yellow with sufficient reporting that were used in the meta-analysis.



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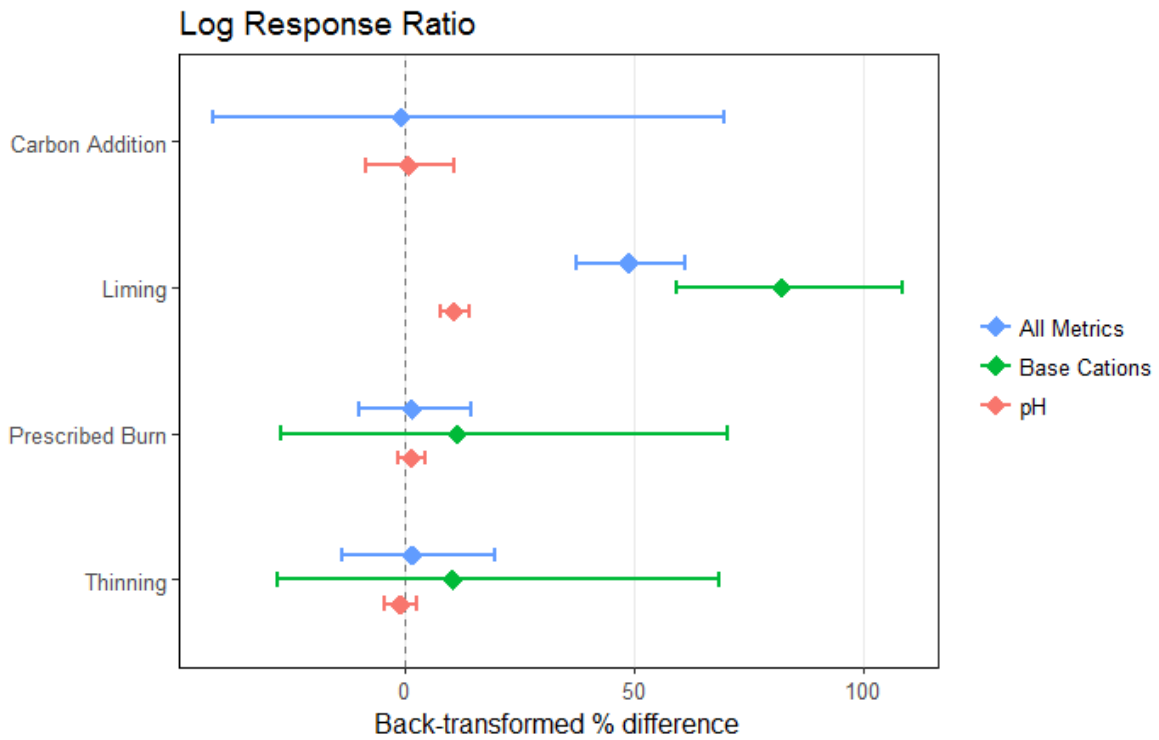
958 **Figure 2: Soil N responses.** Shown are the treatment effects (and 95% CIs) in standard deviation units for
 959 each of the four treatments relative to controls from the Standardized Mean Difference analysis.
 960 Responses include all soil N availability metrics combined (blue), extractable N (green), and net rates of
 961 mineralization or nitrification (red). Metrics associated with each analysis are shown in Table S4 and
 962 effects from individual studies are shown in Figure S1.



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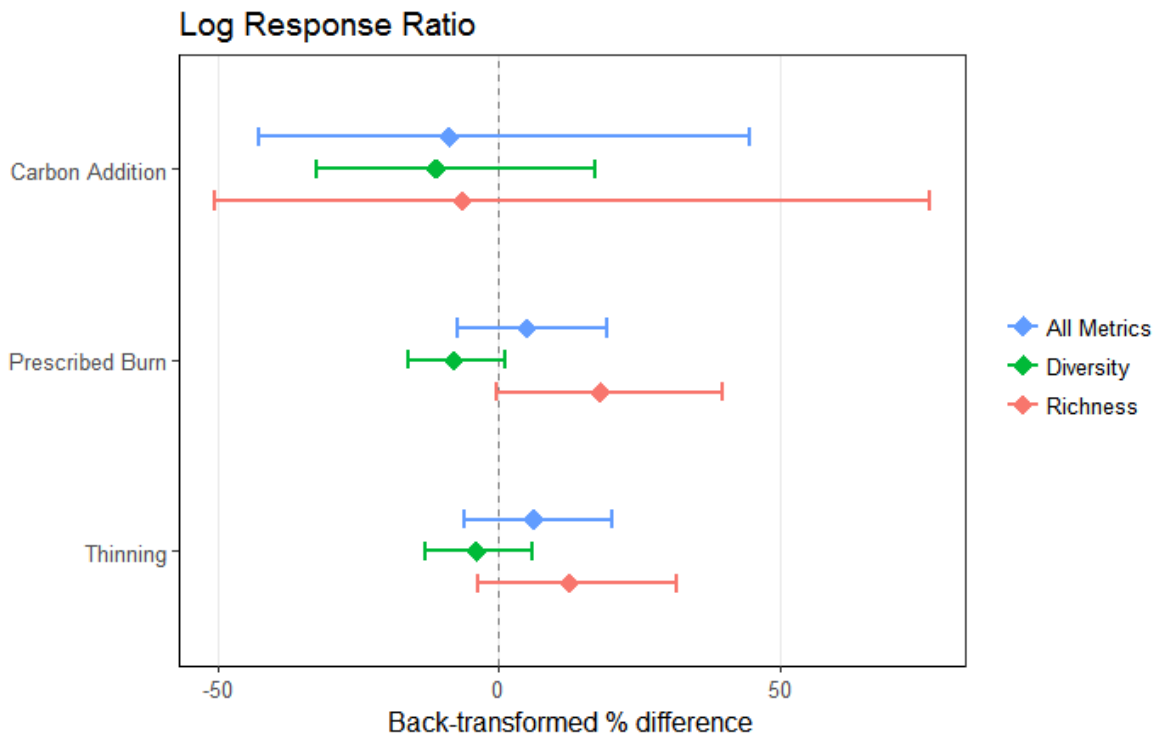
965 **Figure 3: Soil alkalinity responses.** Shown are the back-transformed percent differences (and 95% CIs)
 966 for each of the four treatments relative to controls from the response ratio meta-analysis. Responses
 967 include all soil alkalinity metrics combined (blue), base cations (green), and soil pH (red). Metrics
 968 associated with each analysis are shown in Table S4 and effects from individual studies are shown in
 969 Figure S2. Note that 95% CIs on back-transformed variables may not represent significance accurately,
 970 see Table 2 for statistical tests.



971

972

973 **Figure 4: Tree-plot of treatment effects for plant community response metrics.** Shown are the back-
 974 transformed percent differences (and 95% CIs) for each of the three treatments relative to controls from
 975 the response ratio meta-analysis. Responses include all plant biodiversity metrics combined (blue),
 976 diversity metrics (green: Simpson and Shannon), and species richness (red). Metrics associated with
 977 each analysis are shown in Table S4 and effects from individual studies are shown in Figure S3. There
 978 were no liming studies that examined plant biodiversity. Note that 95% CIs on back-transformed
 979 variables may not represent significance accurately, see Table 2 for statistical tests.



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981

982 Supplementary Material

983 Supplemental Tables

984 **Supplemental Table 1: Keywords used.** All combinations of the keywords below were searched using
 985 Web of Science (October 2017). The query had to include one entry from each category (i.e.,
 986 management type, ecosystem type, nutrient). The titles of these were then read for relevance.

MANAGEMENT METHOD	+	ECOSYSTEM TYPE	+	NUTRIENT
Burn		Forest		Nitrogen
C addition				Acidification
Carbon Addition, C addition				Eutrophication
Cutting				
Fire				
Grazing				
Harvest				
Liming				
Management				
Mitigation				
Mowing				
Recovery				
Remediation				
Restoration				
Seed Addition				
Sod Removal				
Topsoil Removal				
Turf Stripping				

987

988 Supplemental Table S2: Summary of the 29 references used in the meta-analysis. An “X” indicates that either that treatment or response

989 category was examined.

990

Reference	Study Area	Treatment				Response category			Effect sizes
		Carbon addition	Liming	Prescribed burning	Thinning	Soil N	Alkalinity	Plant Biodiversity	
Alcaniz et al. 2016	Montgrí Massif, Catalonia, Spain			X			X		3
Ares et al. 2010	Keel Mountain, Oregon, USA				X			X	30
Binkley et al. 1992	Francis Marion National Forest, South Carolina, USA			X			X		16
Cassidy et al. 2004	Quabbin Reservoir, Massachusetts, USA	X	X			X	X		20
Mclver et al. 2016	Blue Mountains, Oregon, USA			X	X	X	X		12
Mclver et al. 2016	Central Appalachian Plateau, Ohio, USA			X	X	X	X	X	20
Mclver et al. 2016	Central Sierra Nevada, California, USA			X	X	X	X		10
Mclver et al. 2016	Florida Coastal Plain, Florida, USA			X				X	4
Mclver et al. 2016	Gulf Coastal Plain, Alabama, USA			X	X	X	X	X	24
Mclver et al. 2016	Northeastern Cascades, Washington, USA			X	X			X	12
Mclver et al. 2016	Northern Rocky Mountains, Montana, USA			X	X	X	X	X	40
Mclver et al. 2016	Southeastern Piedmont, South Carolina, USA			X	X	X	X	X	28
Mclver et al.	Southern Appalachian Mts.,			X	X	X	X	X	17

2016	North Carolina, USA								
Mclver et al. 2016	Southern Cascades, California, USA			X	X	X	X	X	21
Mclver et al. 2016	Southern Sierra Nevada, California, USA			X		X	X	X	19
Mclver et al. 2016	Southwestern Plateau, Arizona, USA			X	X	X	X	X	12
Ficken and Wright 2017	Fort Bragg Military Reservation, North Carolina, USA			X		X			6
Fisk et al. 2006	Hubbard Brook Experimental Forest, New Hampshire, USA		X				X		4
Ganzlin et al. 2016	Lubrecht Experimental Forest, Montana, USA			X	X	X			14
Hawley et al. 2006	Hubbard Brook Experimental Forest, New Hampshire, USA		X				X		1
Homan et al. 2016	Honnedaga Lake, New York, USA		X				X		4
Johnson et al. 2014	Hubbard Brook Experimental Forest, New Hampshire, USA		X				X		36
Koorem et al. 2012	Koeru, Estonia	X						X	2
Lofgren et al. 2009	Bråtängsbäcken, Gårdsjön IM, Lommabäcken, and Ringsmobäcken, Sweden		X				X		8
Quimet and Moore 2015	Montmorency Experimental Forest, Quebec, Canada		X				X		30
Quimet et al. 2017	Duchesnay, Quebec, Canada		X				X		4
Rizvi et al. 2012	Vosges Mountains, France		X				X		4
Wolk and Rocca 2009	Heil Valley Ranch, Colorado, USA				X			X	4
Wright and Hart 1997	Fort Valley Experimental Forest, Arizona, USA			X		X			3
	Sum	2	9	17	13	14	21	13	408

992 Supplemental Table S3: Full Data Table. See EPA's Environmental Data Gateway for the full dataset
993 (<https://edg.epa.gov>).

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994 Supplemental Table S4: Binning of individual metrics into different analyses.

Response category	Metric	Soil N Availability			Soil alkalinity			Plant Community Diversity		
		All	Extractable N	Net nitrification /mineralization	All	pH	Base cations	All	Species richness	Plant diversity
Soil N availability	Ammonium	X	X							
	inorganic N	X	X							
	mineralizable N	X								
	N mineralization	X		X						
	net ammonification	X		X						
	nitrate	X	X							
	nitrate N	X	X							
	nitrification	X		X						
	net nitrification	X		X						
net N mineralization	X		X							
Soil alkalinity	pH				X	X				
	base saturation				X		X			
	exchangeable Ca				X		X			
	exchangeable K				X		X			
	exchangeable Mg				X		X			
	Ca:Al				X		X			
Plant community diversity	species richness							X	X	
	shannon diversity							X		X
	simpson diversity							X		X

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997 Supplemental Table 5: Full statistical results (see separate file).

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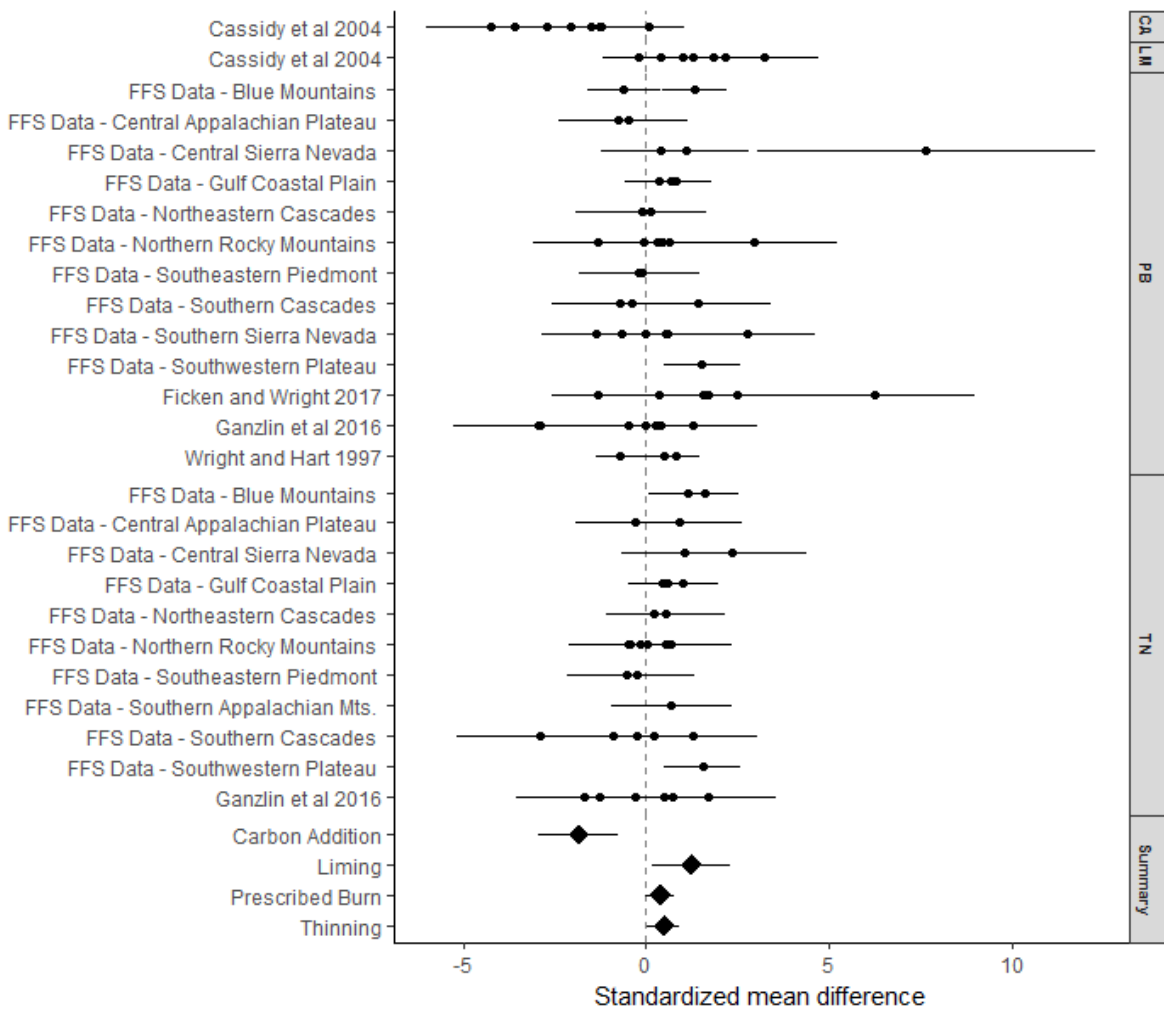
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1001 Supplemental Figures

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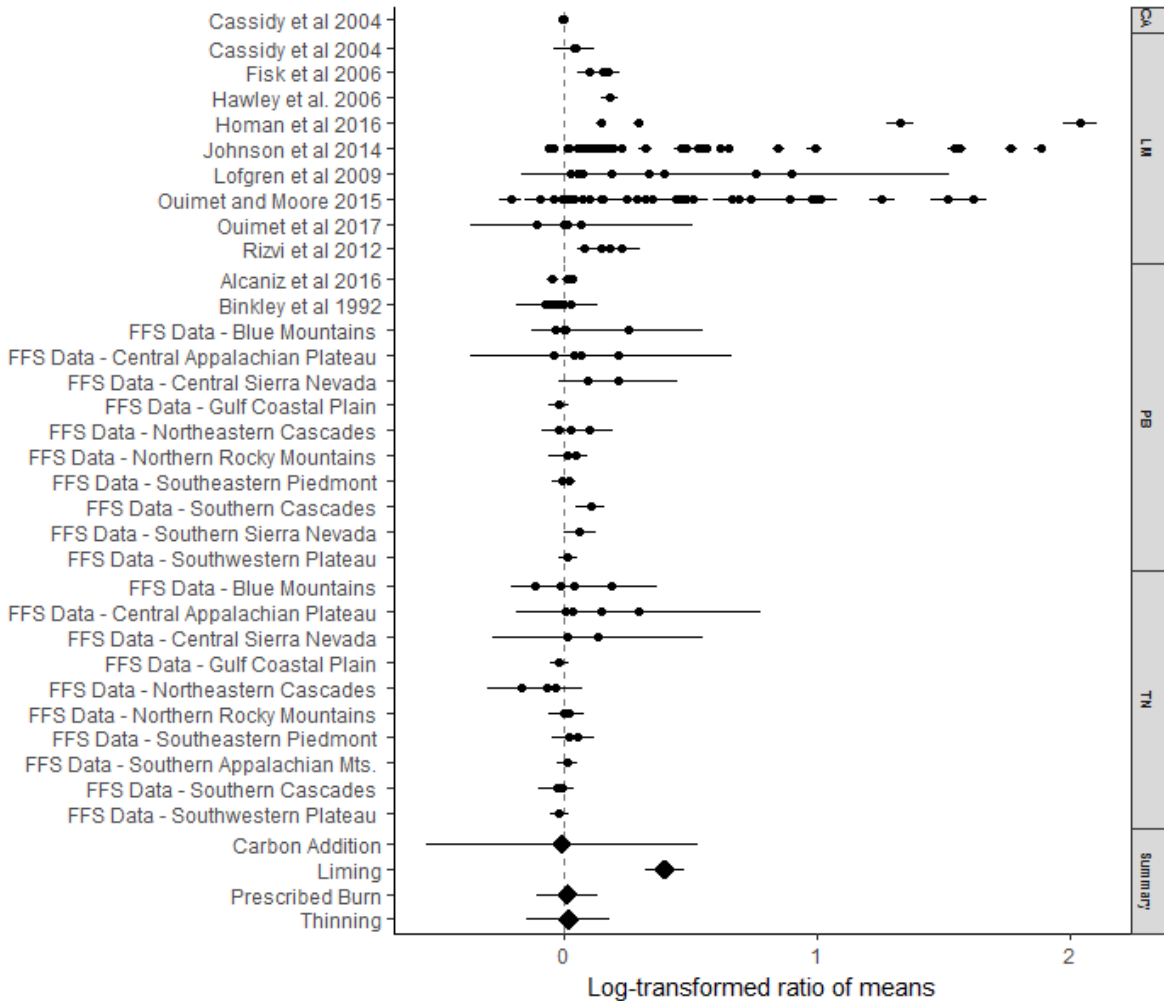
1003 Figure S1: Tree-diagram of soil nitrogen availability responses to carbon addition (CA), liming (LM), prescribed burning
 1004 (PB) and thinning (TN) using standardized mean difference (SMD). Points to the right indicate more soil available N
 1005 relative to controls. Shown below are the effect metric means (points) and standard deviation (bars), from each study,
 1006 and the overall effect of each treatment.



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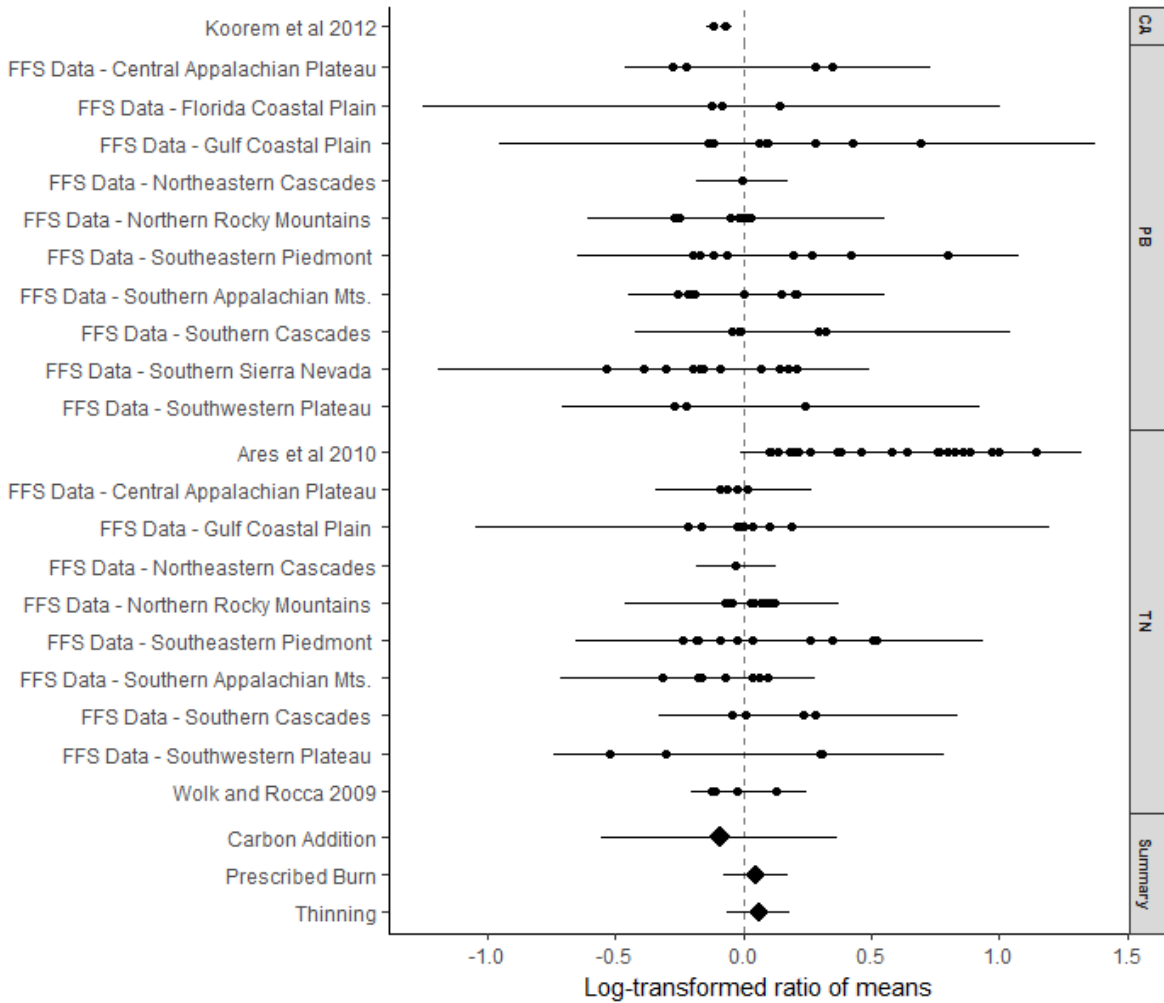
1008

1009 Figure S2: Tree-diagram of soil alkalinity responses to carbon addition (CA), liming (LM), prescribed burning (PB) and
 1010 thinning (TN) using log-transformed response ratios. Points to the right indicate more soil alkalinity relative to controls.
 1011 Shown below are the effect metric means (points) and standard deviation (bars), from each study, and the overall effect
 1012 of each treatment.



1013

1014 Figure S3: Tree-diagram of plant community diversity responses to carbon addition (CA), prescribed burning (PB) and
 1015 thinning (TN) using log-transformed response ratios. Points to the right indicate more plant biodiversity relative to
 1016 controls. Shown below are the effect metric means (points) and standard deviation (bars), from each study, and the
 1017 overall effect of each treatment.



1018

Highlights:

- No single factor improved forest soil N availability, alkalinity, and biodiversity.
- Only carbon addition reduced N availability, only liming increased alkalinity.
- Prescribed burning and thinning had weak or undesirable effects on responses.
- Results are not consistently reported in the literature to support meta-analysis.
- Several treatments may be needed to promote forest recovery from N deposition.