Assessing the impact of human trampling on vegetation: a systematic review and meta-analysis of experimental evidence

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1. Introduction

Recreational pressure can cause many problems for managers of nature reserves, countryside and wilderness (Leung & Marion 2000). Effective management is essential if the recreational usage of natural habitats is to be balanced with the retention of the nature conservation value of a site. In this context, biodiversity managers and researchers in the UK have identified the need for increased knowledge about the impact of recreational activities on biodiversity as one of the 100 most policy-relevant ecological questions (Sutherland et al. 2006); recent differences of opinion regarding open access policies on England's National Nature Reserves underscores the need for robust evidence in this area (Marren 2013). Human trampling, and the response of vegetation to this disturbance, have been a main focus of research on sustainable use of natural habitats for recreation, and have been investigated in many different habitat types around the world. Studies investigating vegetation responses to trampling impacts have utilised various methodologies, including descriptive surveys, site comparisons, before-after control-impact (BACI) designs, and experimental approaches (Sun & Walsh 1998). A standard randomised, controlled experimental design has been described by Cole & Bayfield (1993), and is often used by workers in this field. Standardised procedures should allow for greater comparability between studies, especially

26 where scale-dependent variables affect the measurement of impacts significantly (Taylor, Reader,

27 & Larson 1993). Work on the impact of trampling on vegetation has been reviewed on several

28 occasions (Liddle 1975a; Kuss 1986; Yorks et al. 1997); however, for a transparent and

29 comprehensive synthesis of the available evidence, a systematic methodology should be

30 employed for the retrieval, critical appraisal and pooling of studies (Pullin & Knight 2003;

31 Sutherland et al. 2004; Pullin & Stewart 2006; Stewart 2010).

32 The relevance of trampling studies for conservation managers and practitioners depends on the

33 nature of the managed site, the plant communities contained within, and the type of access in use

34 or being considered (Burden & Randerson 1972; Cole 1987). Some sites may be essentially open

35 access, whilst others may guide or restrict users to paths or delimited areas. Workers studying

36 trampling have divided community responses to trampling into various categories and series with

37 the intention of producing indicators or indices representing the responses of plant communities

38 (Leung & Marion 2000). Resistance, the intrinsic capacity of vegetation to withstand the direct

39 effect of trampling (Liddle 1975b), and resilience, the intrinsic capacity of vegetation to recover

40 from trampling (Kuss & Hall 1991), are most often used as indicators of impact. They allow

41 ecological data to be distilled into categories with biological relevance and conceptually

42 straightforward links to management practice.

43 Studies on trampling have examined the impacts on physiological & morphological vegetation

44 characteristics (Kuss & Graefe 1985), soil fauna (Chappell et al. 1971) and a range of edaphic

45 variables (e.g. Andersen 1995; Ros et al. 2004). However, the response reported most frequently

46 is vegetation cover, which can be used to quantify the vulnerability of vegetation types using

47 measures of resistance and resilience (Cole & Bayfield 1993). Primary studies often present data

48 on vegetation cover as 'relative vegetation cover' (RVC); this is the cover on a trampled plot

49   relative to its initial cover, adjusted for changes in cover on control plots during an experiment

50   (supplemental material 1; Cole & Bayfield 1993).

51   The responses of vegetation to trampling have been reported to be affected by trampling intensity

52   (number of human trampling passes; e.g. Cole 1987, 1995a), frequency (trampling passes per

53   time period; Cole & Monz 2002), distribution (whether trampling passes are dispersed or

54   clumped for a particular trampling frequency; Gallet, Lemauviel, & Rozé 2004), season (Gallet &

55   Rozé 2002), weather (Gallet & Roze 2001), habitat (Liddle 1975b), species (Gallet et al. 2004),

56   Raunkiaer life-form (i.e. perennating bud position) and growth-form (Cole 1995b), and soil type

57   (Talbot, Turton, & Graham 2003). Here, we consider variation in trampling intensity, vegetation

58   resistance, recovery time, Raunkiaer life-form of the community dominant and broad habitat type

59   as potential reasons for heterogeneity in experimental results across primary studies.

60   To our knowledge, no attempt has been made at a formal systematic evaluation of the effect of

61   trampling on vegetation, or at a meta-analytical synthesis of available data. Therefore, the aims of

62   this study were to systematically assess and review the evidence for the effects of human

63   trampling on plant communities, to synthesise experimental data via an appropriate meta-

64   analytical technique, and to investigate variables associated with significant variation in study

65   outcomes. Clearly, robust synthesis relies on robust data, and much relevant data on the effects of

66   human trampling on plants may not be suitable for quantitative synthesis (Yorks et al. 1997). The

67   nature of the available data constrained the specific questions that we were able to address to: (1)

68   Does resilience (i.e. vegetation recovery) differ with respect to trampling intensity, initial

69   resistance of the vegetation, recovery time, Raunkiaer life-form of the community dominant, or

70   habitat? And, (2), how is resilience affected by these covariates and their interactions? We also

71 present tables and lists of relevant primary studies to promote the future synthesis of the

72 considerable amount of observational and mensurative ecological work performed in this area.

73 Practical implications and guidance may be able to be derived where the relationships between

74 resilience and covariates (e.g. trampling intensity or time allowed for recovery) can be

75 manipulated by management decisions. We also comment on the limitations imposed on this

76 systematic review by the original papers, and assess the utility of experimental trampling studies

77 for improving the scientific basis of the management of human trampling impacts in areas of

78 conservation importance. Given the large number of studies that have now investigated the

79 impact of human trampling on plant communities (Yorks et al. 1997; supplemental materials 3

80 and 4), it is important to assess whether the accumulated data available in the primary scientific

81 literature can be effectively mined for ecological patterns providing reliable across-study

82 evidence that can be used to support conservation management decisions (Sutherland et al. 2004;

83 Pullin & Stewart 2006).

84 **2. Methods**

85 *2.1 Identification of relevant studies*

86 Our methodology follows the approach of Pullin & Stewart (2006). Relevant studies were

87 identified via systematic searches of electronic databases, including: JSTOR, ISI Web of

88 Knowledge, ScienceDirect, Google Scholar, DOAJ, Copac, Scirus and Agricola. English

89 language search terms reflecting elements of the review question were used in mining the

90 databases. The references of relevant articles were hand-searched for further studies. To be

91 included in the meta-analysis, a study had to meet the following criteria: (1) The subject of the

92 study must include plant species or assemblages; (2) the experimental treatment must consist of

93 human trampling; (3) the study outcomes must include relative vegetation cover (RVC), or allow

94 this metric to be derived; (4) the study must include controls and replication in randomized

95 experimental designs linking cause and effect, with high strength of inference to non-study

96 situations (Cole & Bayfield 1993; Sun & Walsh 1998). Studies which only met criteria (1) and

97 (2) were also retrieved and summarised (supplemental materials 3 & 4). Articles were assessed

98 for relevance by one reviewer reading article titles and abstracts; a random subset of 40 from 304

99 identified articles was independently assessed for relevance by a second reviewer. Cohen's Kappa

100 test showed the selection of relevant articles to have high inter-reviewer agreement ($K = 0.818$).

101 *2.2 Data extraction and analysis*

102 One reviewer extracted data from graphs or tables; where data were not presented, or could not

103 be derived, attempts were made to contact authors for the original data. Data retrieved from

104 authors within twelve months of the enquiries being made were included in the meta-analysis. In

105 one instance, data were converted from absolute cover to RVC (Ikeda 2003). Due to the

106 presentation of vegetation cover data as RVC, certain aspects of the effect of trampling on

107 vegetation could not be directly investigated. Effect sizes used in meta-analysis usually

108 summarise the effect of a treatment or intervention by comparing treatment and control means

109 and variability before and after the treatment in question (Egger, Smith, & Altman 2001).

110 However, data presented as RVC are relative to mean initial vegetation cover: measures of initial

111 variability prior to trampling, necessary for effect size calculation, are not generally presented by

112 the authors of primary studies. This means that the immediate post-trampling impact on

113 vegetation (a typical measure of resistance) cannot be used as the dependent variable in meta-

114 analyses, because appropriate data summarising the pre-trampling state of the vegetation cannot

115 be extracted from the RVC metric. Therefore, we investigated differences in RVC between the

116 period immediately after trampling and the time-point farthest from the trampling application

117 recorded in each study, thus increasing the predictive power and increasing independence where

118   there was a choice of time ranges in a study. Because primary studies often investigated more

119   than one habitat, trampled at more than one intensity, single effect size estimates (hereafter

120   referred to as 'trials') were extracted for any given habitat and trampling intensity pair within a

121   single study. Investigating the presence of a recovery period, irrespective of length, as the main

122   effect also increased the independence of trials within studies, giving each trial its own

123   comparator rather than comparing several trampling intensities to a single control. Thus, as other

124   reviewers have found practical (Yorks et al. 1997), we investigated vegetation recovery as the

125   dependent variable on a per trial basis, and defined this as the resilience of the vegetation. It

126   should be noted that the terms resilience and resistance have been used elsewhere to refer to

127   indices that combine data on both vegetation response and trampling intensity to produce a metric

128   estimating what has been defined as the 'vulnerability' of any given site or vegetation type (Cole

129   & Bayfield 1993).

130   The main effect of the presence of a recovery period on trampled vegetation, calculated as the

131   mean RVC at the final monitoring point of all replicates within a trial, minus the mean RVC

132   immediately after trampling for the replicates, and other reasons for variability in vegetation

133   recovery (trampling intensity, resistance, length of recovery period, Raunkiaer life-form, and

134   habitat) were explored using meta-analysis and meta-regression (Deeks, Altman, & Bradburn

135   2001; Gurevitch & Hedges 2001). Cohen's d effect sizes (Deeks et al. 2001) representing the

136   change in vegetation cover between initial post-trampling monitoring and the final monitoring

137   time-point were derived from the two RVC means with standard deviations and sample sizes

138   (where the sample size is the number of experimental replicates, not the number of sub-sampled

139   plots within replicates). Cohen's d uses a pooled estimate of standard deviation, allowing for the

140   paired nature of the data points. Data were pooled and combined across trials using DerSimonian

141   & Laird random effects meta-analysis based on standardised mean difference (SMD;

142 DerSimonian & Laird 1986; Cooper & Hedges 1994). Meta-analysis combines the main effects

143 from individual trials into a single estimate, whilst also taking the precision of each estimate into

144 account (Gurevitch & Hedges 2001). Meta-analysis also increases statistical power and allows

145 the quantification and, where possible, exploration of variation between trials (Deeks et al. 2001;

146 Gurevitch & Hedges 2001). The random effects model assumes that there is variation amongst

147 the true trial effects, and the aim of the analysis is to quantify such variation in the effect

148 parameters; it is therefore appropriate for ecological questions where the true effect is likely to

149 vary between trials (Gurevitch & Hedges 2001; Stewart 2010).

150 The effect of a post-trampling recovery period on plant growth as quantified by changes in RVC

151 (i.e. vegetation resilience) was examined via the visual inspection of the forest plot of the

152 estimated main effects from the trials, along with their 95% confidence intervals, and by formal

153 tests of heterogeneity undertaken prior to meta-analysis (Thompson & Sharp 1999). Publication

154 bias was investigated by examination of funnel plot asymmetry and the Egger test (Egger et al.

155 1997). The relationships between the effect of a recovery period and the explanatory variables

156 (length of recovery period, resistance and trampling intensity) were tested using random effects

157 SMD meta-regressions in Stata v. 8.2 (Stata Corporation 2003) using the program Metareg (Sharp

158 1998). Meta-regression investigates the explanatory power of covariates for the observed pattern

159 of main effect sizes; the random effects model acknowledges the potential for residual

160 heterogeneity not explained by the covariate(s) between trials, therefore corresponding to random

161 effects meta-analysis in assuming that the between-trial variance is not zero (Thompson &

162 Higgins 2002). Meta-regressions were limited to continuous trial-level variables taking a range of

163 values, and, to avoid data-dredging, were specified *a priori* (Thompson & Higgins 2002) as

164 resistance, length of the recovery period, and trampling intensity. Resistance was coded for meta-

165 regressions as the mean RVC across replicates for each trial immediately after trampling had been

166 applied. Collinearity between independent variables was investigated prior to performing the

167 multiple meta-regressions (Zuur, Ieno, & Elphick 2010).

168 The use of subgroup analyses allows the investigation of variation in the main effect amongst

169 particular groups of trials that are hypothesised to be biologically significant (Brookes et al.

170 2001). We used subgroup analyses to explore variation in the effect of a recovery period amongst

171 plant communities of different Raunkiaer life-forms and different habitats. Raunkiaer life-form

172 subgroups, delineated by the position of the perennating buds (Kent 2012), were: (juvenile)

173 phanerophytes; chamaephytes; hemicryptophytes; geophytes; helophytes (where vulnerable to

174 trampling); and therophytes. The life-form category of the plant species with the highest mean

175 percentage cover, i.e. the community dominant, was taken as the category for a trial. Broad

176 habitat categories were: alpine or tundra; temperate coniferous forest; temperate deciduous forest;

177 subalpine and montane grass or shrubland; temperate shrubland; and temperate grassland. *Post*

178 *hoc* within-subgroup multiple meta-regressions were also used to investigate the differential

179 explanatory power of our *a priori* potential effect modifiers (resistance, length of the recovery

180 period, and trampling intensity) for each life-form; collinearity between independent variables

181 was also checked for all within-subgroup multiple meta-regressions (Zuur et al. 2010).

182 To investigate the potential effects of non-independence between trials on our results, due to, for

183 example, individual studies conducting multiple trials testing different trampling intensities in

184 similar habitat types, we also performed sensitivity analyses for all meta-regressions using robust

185 variance estimation (Hedges, Tipton, & Johnson 2010). Robust variance estimation was

186 performed according to (Hedges et al. 2010) on DerSimonian & Laird random effects meta-

187 regressions conducted using the metafor package (Viechtbauer 2010) for the statistical software R

188 v. 3.0.2 (R Core Team 2005).

3. Results

3.1 Systematic search results

Searching and retrieval were conducted between October 2005 and July 2006. Three hundred and four articles were judged relevant at the title level; 145 articles remained in the systematic review after the title and abstract filter stage. Of the 145 only 24 were randomised controlled experiments eligible for inclusion in the final meta-analysis. Of these 24, nine presented data that could be extracted, or which were provided by the authors (Table 1). These nine contributed 188 trials to the meta-analysis. A methodological overview of articles not included in the meta-analysis, but which used a comparator or control in their experimental design, is provided in supplemental material 3. Lists of studies not using a comparator or control, and of those studies which could not be retrieved within the resource constraints of the project, are also provided for the benefit of future reviewers (supplemental material 4).

3.2 Meta-analyses

3.2.1 Post-trampling vegetation recovery

The 95% confidence intervals (horizontal arms) of the 188 trial effect sizes estimating post-trampling recovery of vegetation cover (resilience), via changes in RVC, are shown in a forest plot (Fig. 1); the weighted central point estimates have been omitted for clarity. An enlargeable PDF version of Figure 1 with trial identifiers, trampling intensities and Cohen's *d* effect sizes (with 95% confidence intervals) is also provided for closer inspection (supplemental material 2). The broken vertical line shows a positive, significant change in RVC after a period of recovery across all trials (SMD = 1.357, z = 11.13, $p < 0.001$; Fig. 1). The range of variation in the contributing trials gave significant variation in effect size ($\chi^2 = 576.83$, d.f. = 187, $p < 0.001$), with trials located both sides of the line of no effect (no recovery), and with some trials exhibiting

212 wide confidence intervals. Funnel plot asymmetry (Fig. 4) and the Egger test (Egger et al. 1997)

213 suggest that there is potential bias in the data set, as there are fewer small negative trials than

214 would be expected by chance (Egger bias = 4.7, $p < 0.001$).

215 **3.2.2 Multiple meta-regressions and subgroup analyses: effects of covariates**

216 The multiple meta-regression found that significant heterogeneity in vegetation recovery was

217 explained by initial vegetation resistance (coeff. = -0.357, $z = -8.77$, $p < 0.001$), and the length of

218 the recovery period (coeff. = 0.246, $z = 2.00$, $p < 0.045$); however, there was less evidence for an

219 effect of trampling intensity (coeff. = -0.001, $z = -1.74$, $p = 0.082$). Correlations (Pearson's r)

220 between independent variables were all below 0.35.

221 Subgroup analyses were conducted on Raunkiaer life-form categories based on the dominant

222 field layer vegetation of each trial (Fig. 2). Vegetation dominated by phanerophytes ($n = 40$; SMD

223 = -0.033, $z = 0.63$, $p = 0.528$), chamaephytes ($n = 47$; SMD = 0.288, $z = 1.15$, $p = 0.248$),

224 helophytes ($n = 4$; SMD = 1.918, $z = 1.63$, $p = 0.102$) or therophytes ($n = 4$; SMD = -0.737, $z =$

225 1.71, $p = 0.087$), did not show a significant effect of the presence of a recovery period on RVC

226 (95 % CIs cross zero; Fig. 2); however, all of these life-form groupings displayed significant

227 within-subgroup heterogeneity ($p < 0.01$). Hemicryptophytes ($n = 72$; SMD = 1.955, $z = 12.59$, p

228 < 0.001) and geophytes ($n = 21$; SMD = 1.660, $z = 4.59$, $p < 0.001$) both showed a positive,

229 significant main effect of a recovery period on RVC (95 % CIs do not cross zero; Fig. 2), but also

230 displayed significant heterogeneity ($p < 0.01$).

231 For chamaephytes, hemicryptophytes and geophytes, *post hoc* within-group meta-regressions

232 examined the effects of resistance, length of a recovery period and trampling intensity on

233 vegetation recovery. Across the meta-regressions, correlations (Pearson's r) between independent

234   variables were all below 0.52. The meta-regressions found the initial resistance of the vegetation

235   to have a small but strong negative correlation with resilience for chamaephytes and

236   hemicryptophytes (Table 2); the length of the recovery period was less important, but positive for

237   chamaephytes and negative for hemicryptophytes (Table 2). Recovery time could not be included

238   in the geophyte meta-regression due to all data points being reported one year after trampling;

239   however, resistance was found to have a small, but significant, negative correlation for geophytes

240   (Table 2). Trampling intensity was non-significant for chamaephytes, hemicryptophytes and

241   geophytes (Table 2); the residual variation was significant for all three life-forms (chamaephytes:

242    $p = 0.044$; hemicryptophytes: $p < 0.001$; geophytes: $p = 0.001$). Simple meta-regressions for

243   trampling intensity within these subgroups also found no effect for hemicryptophytes ($p = 0.638$)

244   and chamaephytes ($p = 0.883$), suggesting that the finding of no effect of trampling intensity was

245   not being confounded by other covariates; however, trampling intensity had a significant effect

246   on geophytes ($p = 0.001$), suggesting a stronger relationship (i.e. confounding variation) between

247   trampling intensity and vegetation resistance in this subgroup. Habitat-based subgroup analyses

248   revealed positive and significant main effects ($p < 0.05$; Fig. 3) and significant heterogeneity ($p <$

249   0.01) for all habitats, except temperate grassland, for which power was very low ($n = 4$).

250   **3.2.3. Sensitivity analyses**

251   The use of robust variance estimation to account for the potential non-independence of trials

252   within studies reduced the significance of all covariates tested. **For the main multiple meta-**

253   **regression**, initial vegetation resistance (coeff. = -0.039; t = -8.16, $p < 0.001$) remained

254   significant, whilst both length of the recovery period (coeff. = 0.262; t = 1.80, $p = 0.13$) and

255   trampling intensity (coeff. = -0.001; t = -1.38, $p = 0.226$) became non-significant. For the

256   Raunkiaer subgroup analyses, hemicryptophytes were the only group for which a significant

257   effect of the presence of a recovery period on RVC remained after adjustment for potential trial

258 non-independence. The effects of robust variance estimation on the Raunkiaer life-form within-

259 subgroup meta-regressions could only be investigated for chamaephytes; robust standard errors

260 for the hemicryptohyte and geophyte subgroups could not be produced due to the small number

261 of studies involved (4 and 3 respectively; Hedges et al. 2010). The effect of robust standard errors

262 on the chamaephyte multiple meta-regression was to make all covariates non-significant:

263 resistance: coeff. = -0.033; t = -2.60, p = 0.122; recovery time: coeff. = 0.478; t = 1.18, p = 0.359;

264 trampling intensity: coeff. = -0.001; t = -0.49, p = 0.672.

265 **4. Discussion**

266 *4.1 Meta-analyses*

267 We have found that, over the human trampling studies synthesized, vegetation generally

268 recovered to some extent: there is an average significant positive effect of the presence of a

269 recovery period on relative vegetation cover (RVC). However there is also significant

270 heterogeneity, exceptions to this trend are frequent, and the effect may be over-estimated as a

271 result of bias. Our results suggest that the initial resistance of a plant community, and the length

272 of the recovery period, may be better predictors of vegetation resilience than the intensity of

273 trampling undergone; that is, intrinsic properties of vegetation appear to be some of the most

274 important determinants of resilience, with the magnitude of the actual disturbance explaining

275 much less of the community response. The absence of a relationship between resilience and

276 trampling intensity within the main meta-regression may be due to confounding variation caused

277 by the differential vulnerability of different vegetation types, therefore these results are equivocal.

278 However, the fact that *post hoc* life-form subgroup multiple meta-regressions for chamaephytes

279 (n = 47) and hemicryptophytes (n = 72) also showed a lack of an effect of trampling intensity on

280 resilience, supports the interpretation that intrinsic factors, i.e. plant traits, are often likely to be of

281 primary importance for determining the vulnerability of vegetation to trampling (Cole 1995b).

282 The within-life-form subgroup meta-regressions confirmed that, except possibly for geophytes,

283 life-form type was not obviously confounding the finding of no effect of trampling intensity on

284 resilience within the main analysis. Overall, these results support a situation where particular

285 plant functional traits are likely to be more important than projected intensity of use when

286 considering the siting of recreational activities involving human trampling. This somewhat

287 surprising result has important management ramifications because it suggests that even relatively

288 low intensity trampling could be as damaging as high intensity trampling in certain plant

289 communities. Thus, trampling may sometimes be unsustainable for vulnerable vegetation,

290 potentially creating conflict between even relatively limited access and plant species- or

291 community-focused conservation objectives.

292 The importance of initial vegetation resistance for recovery, indicated by the multiple meta-

293 regressions, is likely to be a partial reflection of the negative correlation expected between

294 resistance and resilience where vegetation is able to recover, and confirms the main effect in

295 showing that recovery is the typical response for the levels of disturbance investigated in the

296 studies included in our meta-analysis. However, the use of the relative metric RVC means that the

297 recovery predicted for an impacted stand of vegetation is dependent on the initial absolute

298 vegetation cover: a stand with high absolute cover and high resistance may not have much

299 potential for recovery (restricted to 100%); this means that our analysis may slightly

300 underestimate the typical recovery of more heavily impacted stands. However, it is arguably the

301 relative resilience of different vegetation communities that is of greatest importance for

302 informing sustainable management. In this respect our subgroup analyses confirmed the

303 importance of Raunkiaer life-form (Cole 1995b) across the 188 trials investigated, suggesting that

304 hemicryptophytes and geophytes will be more resilient to trampling impacts relative to other life-

305 forms. In contrast, chamaephyte-dominated vegetation did not show a main effect of recovery;

306 indeed, chamaephyte-dominated communities have been shown to die-back after trampling

307 disturbance, despite initially high resistance (Cole 1995b; Cole & Monz 2002). The negative

308 relationship between resistance and resilience in the chamaephyte subgroup meta-regression

309 reflects vegetation die-back after initially high resistance, rather than re-growth after initially low

310 resistance. However, the chamaephyte subgroup meta-regression also showed a positive

311 correlation of recovery time with resilience, suggesting that limited recovery may occur after die-

312 back, given a period free from further disturbance. The negative correlation of recovery time for

313 the hemicryptophyte subgroup may indicate that where recovery is not observed in the shorter-

314 term, other factors, changes to soil characteristics for example, could make full recovery less

315 likely.

316 The investigation of habitat as a reason for heterogeneity did not reveal any clear differences

317 between subgroups (Fig. 3); however, the classification system used was broad, and sample sizes

318 were small for some categories. As the vulnerability of vegetation to trampling may be related to

319 primary productivity (Liddle 1975b), a greater correlation between habitat and vegetation

320 response might have been expected. The absence of a relationship in our results may be due to

321 subtle biases relating to the distribution of trampling intensities and the length of recovery

322 periods, confounding with life-form and the small number of replicates per habitat in our dataset.

323 It is also possible that RVC does not reveal differences in production across habitat types as

324 effectively as other measures, such as vegetation height or biomass.

325 *4.2 Critical analysis*

326 We have found that the nature of the primary data available for meta-analytical synthesis of

327 trampling impacts on vegetation requires a cautious approach to interpretation. Because of the

328 lack of any measure of initial cover or variability in the majority of the primary studies analysed,

329 we have focused on the effect of the presence of a period of recovery on post-trampling

330   vegetation change, i.e. resilience. It should be remembered that because RVC is calculated

331   relative to initial vegetation cover, similar RVC changes may not equate to equivalent absolute

332   changes in vegetation cover. Additionally, a non-significant main effect may either represent a

333   lack of vegetation recovery, or a plant community with high resistance and therefore little

334   potential for recovery; this is also a problem with using RVC to measure resilience on a study-

335   level basis (Cole & Bayfield 1993). This problem could be ameliorated by ensuring analyses of

336   resilience refer back to primary data, possibly by investigating the absolute resistance of

337   vegetation as an explanation for heterogeneity in recovery. The availability of raw data in

338   supplementary material or data repositories, or higher response rates from authors of primary

339   studies, would increase the value of trampling studies for synthetic, predictive research in

340   conservation, and for research on the relationship between plant traits and disturbance. The

341   availability of raw data would also allow a more powerful analysis using a hierarchical modeling

342   approach, with increased power to detect real differences between groups, improved ability to

343   explore interaction within and between trials, and would also allow for formal explorations of

344   model choice (Stewart et al. 2012). Hierarchical modelling is also an alternative to robust

345   variance estimation in accounting for the potential non-independence of trials within studies.

346   Our analysis is also limited by the characteristics of the primary studies. Most of the studies

347   investigated several trampling intensities within single habitats, and some habitats and species

348   were over-represented across studies. We therefore had to balance the competing biases of

349   aggregation and non-independence in our synthesis. Non-independence can cause the

350   overestimation of significance levels and the underestimation of confidence intervals (Gurevitch

351   & Hedges 2001); however, pooling effects across dependent and independent variables allows the

352   investigation of interesting ecological heterogeneity, and increases the strength of inference to

353   non-study situations, essentially in the same way as generalizing over subjects within studies

354   (Rosenthal 1991). The sensitivity analysis undertaken here suggests that, if non-independence is a

355   strong property of the data we have analysed, then the most robust conclusions are those for the

356   effects of recovery time and initial vegetation resistance in the main meta-regression.

357   The most important shortcoming exposed by our systematic review is possibly the lack of high-

358   quality experimental information. Clearly, as we have found, restricting accepted information to

359   randomized controlled experiments reduces the proportion of the literature which can be included

360   in a meta-analytic review. This means that certain trends are confounded; for example, Raunkiaer

361   subgroup analyses occasionally resulted in life-forms at certain time-points originating from a

362   single study. Additionally, only one of the admitted studies examined long-term, chronic

363   trampling, and then only over three years (Cole & Monz 2002). This means that our conclusions

364   are less certain for those situations in which chronic trampling impacts affect the physical,

365   chemical and biological properties of the soil, subsequently affecting vegetation growth and

366   succession over a longer period (Burden & Randerson 1972). However, it seems highly likely

367   that negative impacts are certain at high-levels of chronic trampling, and therefore potentially of

368   less importance for evidence-based management seeking to balance biodiversity conservation

369   interest and lower-intensity access to open sites. This is a key question for societies in which

370   open access to sites is becoming more common (Sutherland et al. 2006; Marren 2013). The

371   inclusion of experimental designs with lower internal validity does not necessarily increase

372   external validity, and can simply result in more uncertainty (Stewart 2010). This is a compelling

373   reason for excluding relevant but low quality data from statistical analyses, and reflects the

374   conclusions of other workers who have reviewed trampling impacts (Yorks et al. 1997). However,

375   meta-analytical techniques designed to handle variable quality data are under development, and

376   could prove useful for synthesizing such data in the future, provided that the uncertainty

377 associated with lower-quality methodologies is adequately expressed. Bayesian Belief Networks

378 have been utilised in ecology in such situations (Newton et al. 2007).

379 In attempting this synthesis we have observed that the research methodologies and reporting of

380 existing experimental trampling studies may be inadequate for the underpinning of scientific

381 management of plant species and communities of conservation concern under potential threat

382 from recreational access. Changes in species composition or richness are not routinely reported,

383 meaning that decision-making may be based on vegetation cover indicators, such as RVC, that

384 may not approximate to full ecological recovery to pre-disturbance conditions (Hylgaard 1980).

385 Furthermore, the measure of RVC may be misleading as it lacks information on initial absolute

386 plant cover, which may be of increased importance for the consideration of chronic trampling

387 impacts. Whilst the standardisation of methods reduces unexplained variation, and increases

388 comparability between studies, presenting results in a way which precludes the efficient meta-

389 analysis of important responses is counter-productive to the aims of synthesis; this may be

390 especially true where such synthesis is important for disseminating ecological results to

391 conservation practitioners managing sites with under-studied, or unstudied, vegetation

392 communities. Researchers could address these issues by making raw data available through

393 online supplementary material where they do not present it in manuscripts, and by increased

394 consideration of the potential uses of their results for meta-analytical studies.

395 *4.3 Management Implications*

396 The evidence presented here, systematically accumulated across high-quality experimental

397 studies, suggests that vulnerable vegetation of conservation value should not be trampled,

398 irrespective of the projected intensity of use. The range of trampling intensities investigated in

399 the primary studies synthesized here suggests that even moderate disturbance can have significant

400 effects on plant communities. Simple indicators such as life-form of the community dominant

401 may then be useful for rapid assessments of a community's vulnerability to recreational pressure.

402 Zonation of recreation into high and low intensity usage, with 'honey-pots' located away from

403 vulnerable vegetation, may be a more effective conservation strategy than encouraging

404 moderately intensive but more widespread recreational usage; especially given that occasional

405 use results in the development of informal path-networks which may subsequently encourage

406 further disturbance (Roovers et al. 2004).

407 Sites of conservation importance dominated by phanerophytes, chamaephytes, helophytes or

408 therophytes should not experience regular trampling disturbance if deleterious impacts are to be

409 avoided. Reducing trampling intensities may not be effective where adverse impacts are already

410 occurring, although we did find a negative relationship between initial resistance and resilience

411 for chamaephyte-dominated vegetation (i.e. high initial impacts may be followed by some

412 recovery). Conversely, the current evidence base suggests that vegetation dominated by

413 hemicryptophytes and geophytes, life-forms with more protection for their perennating buds

414 (Kent 2012), recovers to a greater extent than vegetation dominated by other life-forms, and

415 could therefore potentially be trampled more intensively, provided monitoring is undertaken to

416 provide early warning of deterioration or unsustainable use.

417 *4.4 Future Work*

418 Systematic reviews and meta-analyses should be periodically revisited in order to incorporate

419 new data and to test new hypotheses or analytical techniques (Pullin & Stewart 2006; Stewart

420 2010). Since the research reported here was undertaken, several new studies investigating

421 trampling impacts have been reported. In contrast to the work summarised here, Bernhardt-

422 Römermann et al. (2011) found no evidence for an effect of Raunkiaer life-form in their pan-

423   European experimental study of trampling disturbance; whilst other recent studies have found

424   new evidence supporting the importance of this plant trait (Andrés-Abellán et al. 2006).

425   Bernhardt-Römermann et al. (2011) did report evidence for the importance of plant rosette-type,

426   the categories of which used by Bernhardt-Römermann et al. (2011) are classified as

427   hemicryptophyte subtypes (Kent 2012), supporting the importance of perennating bud position

428   and protection for plant responses to trampling disturbance. Increased availability of full plant

429   community data in primary studies could enable future reviewers to estimate the relative

430   abundance of plant traits within vegetation, providing further insights into the importance of

431   Raunkiaer life-form in contrast to other community-weighted plant traits (Violle et al. 2007).

432   Given that the large total number of trampling studies that our systematic review uncovered

433   (supplemental materials 3 and 4) is still almost certainly an underestimate of the evidence-base

434   (due to our only following-up studies referenced within studies to a depth of one remove) we

435   suggest that further investigation of ways to extract the maximum information from published

436   studies will be the most efficient way of confirming when life-form, or other plant functional

437   traits, are likely to be important indicators of vegetation responses to trampling. For example, a

438   risk ratio metric of the relative proportions of different plant functional groups allowed Newton et

439   al. (2009) to summarize a much larger proportion of the primary literature on north-west

440   European heathland management than would have been possible if community composition data

441   were demanded. The increasing availability of global plant trait data (Kattge et al. 2011), suggests

442   that the concepts and information presented here could be extended to test new hypotheses about

443   the relationship between trampling disturbance and plant vulnerability. We suggest that

444   conservation planners, practitioners, and ecologists with an interest in vegetation trampling,

445   should develop a global prospective collaboration to ascertain priority questions and establish

446 standards for monitoring and data reporting. These activities would facilitate future synthesis and

447 maximize the potential for scientific evidence to inform policy in the increasingly important area

448 of research into human impacts on ecosystems.

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Figure 1

Post-trampling recovery of vegetation.

A forest plot of the 188 effect size estimates pooled using random effects meta-analysis. Solid horizontal lines represent trial 95 % confidence intervals. Trial confidence intervals not crossing the zero line of no effect indicate a positive or negative significant effect at the 5 % level. Individual central point estimates are omitted for clarity.

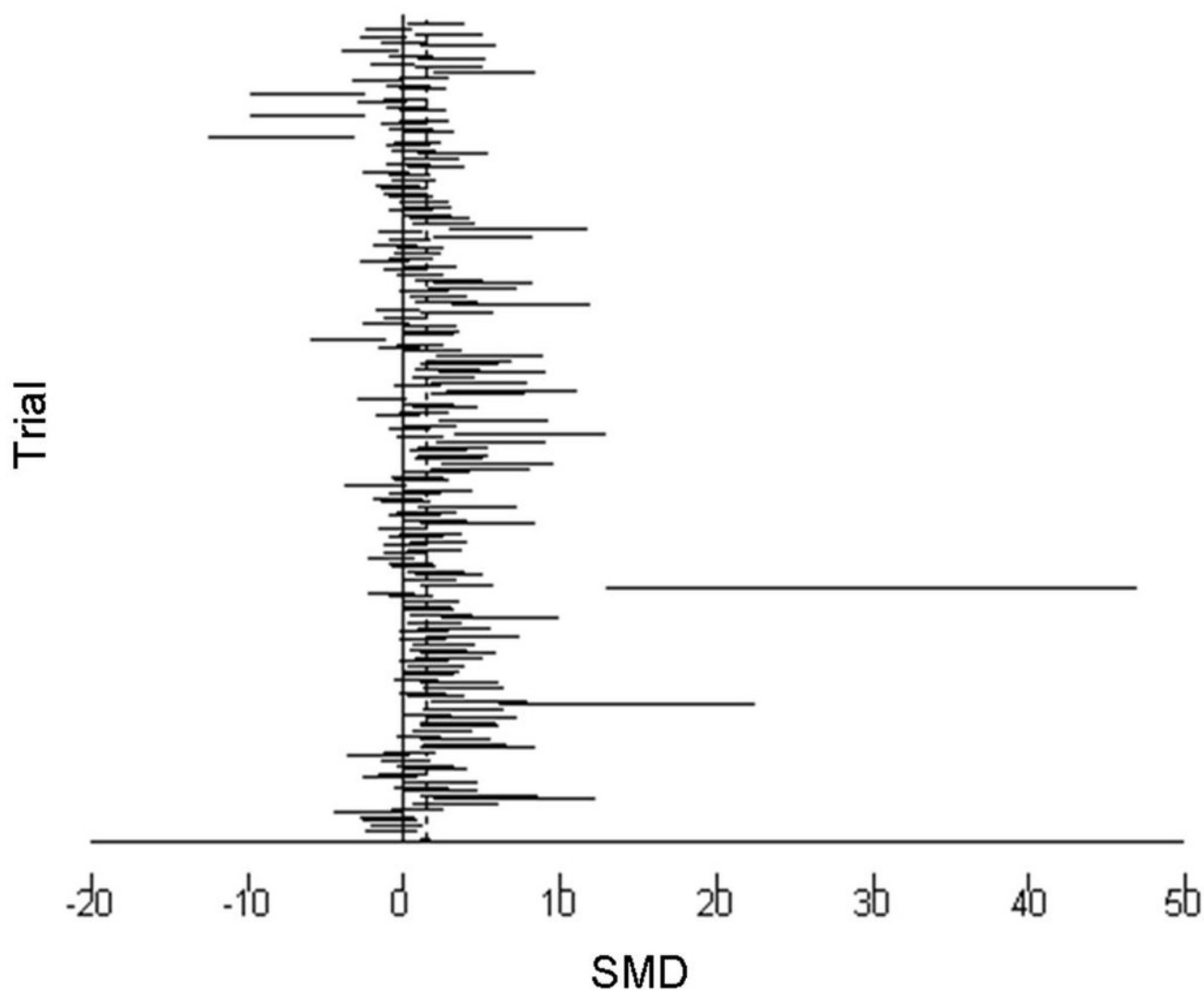



Figure 2

Post-trampling recovery of vegetation for Raunkiaer subgroups.

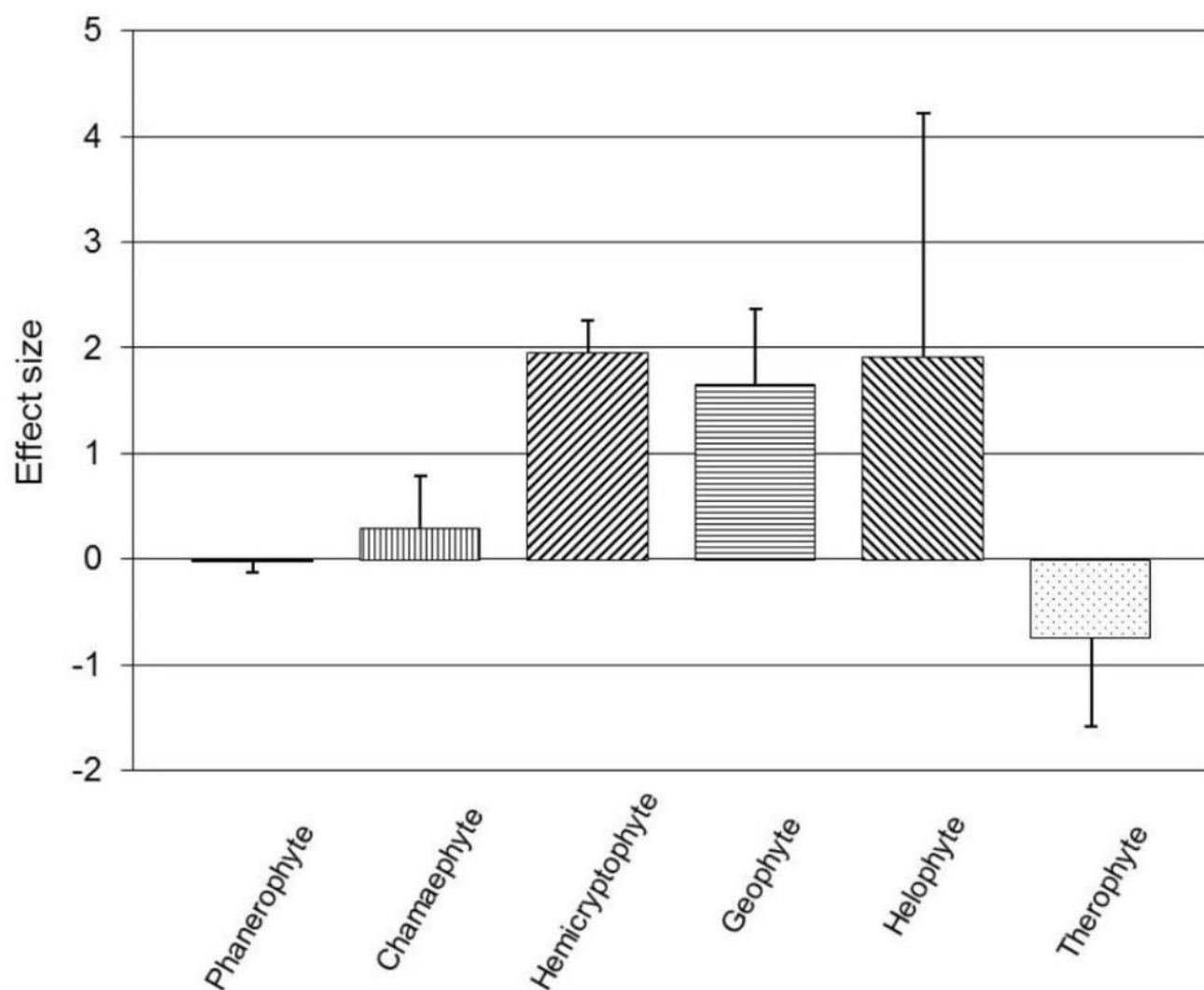
Point estimates of effects sizes with 95 % confidence intervals for Raunkiaer life-form

subgroup analyses.

Figure 3

Post-trampling recovery of vegetation within broad habitat types.

Point estimates of effects sizes with 95 % confidence intervals for habitat subgroup analyses.

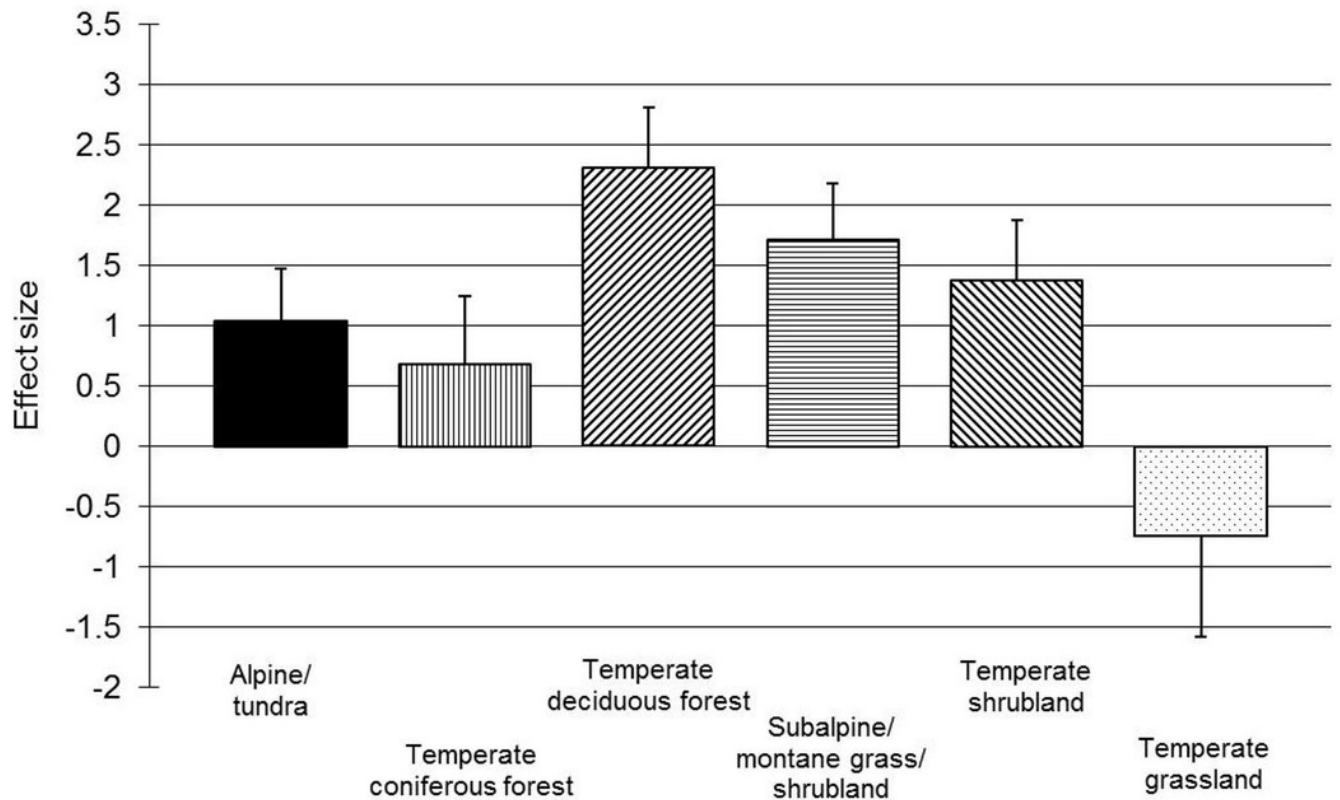



Figure 4

Funnel plot of main effect size estimates.

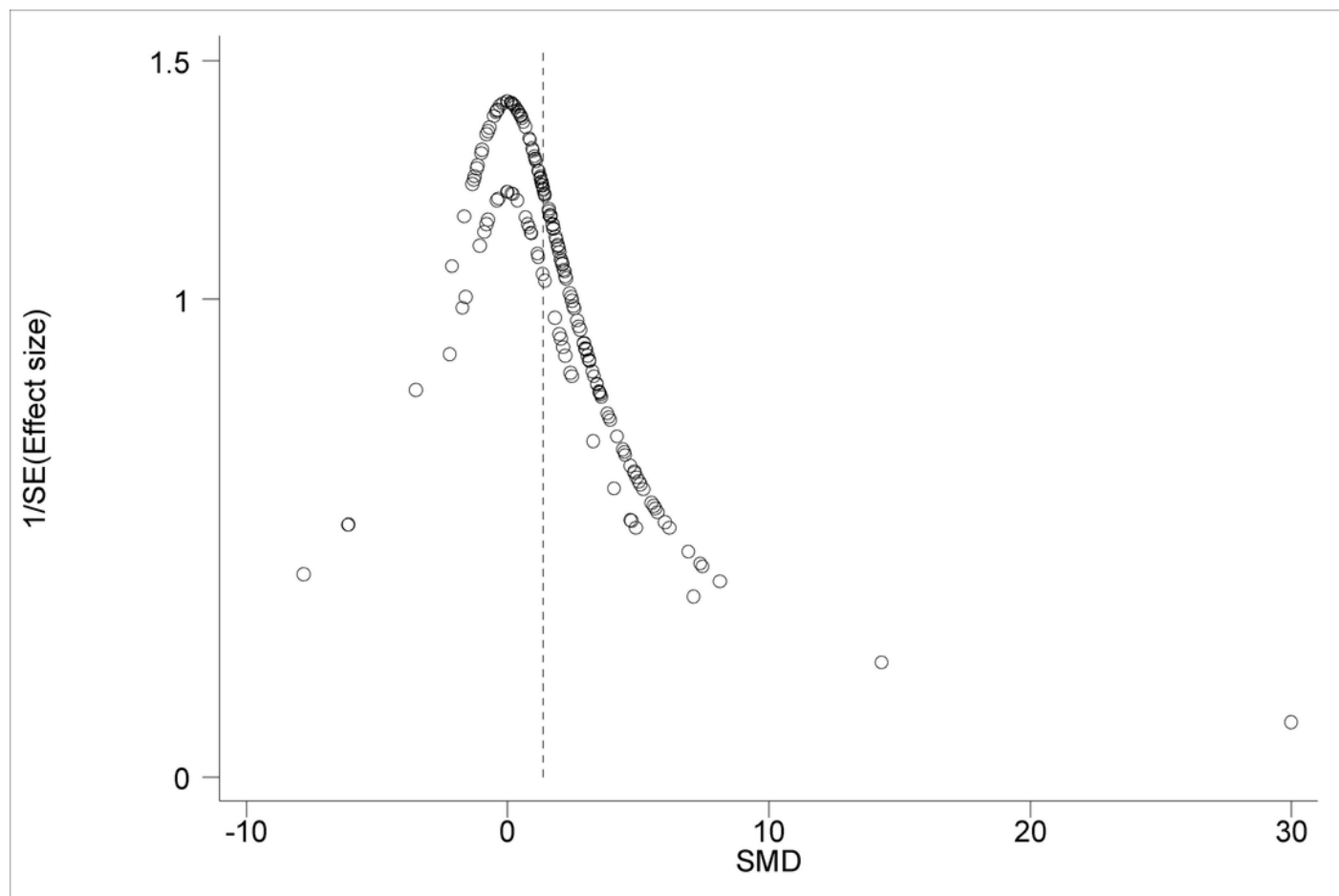
A funnel plot of 188 main effect size estimates in relation to inverse variance. Funnel plot asymmetry illustrates that there are fewer small negative effect sizes than small positive effect sizes indicating a possible publication bias.

Characteristics of studies included in the meta-analysis.

Table 1. Characteristics of studies included in the meta-analysis

Study	Habitat(s)	Trampling intensities (passes) & disturbance type	Trial sample size	Final follow-up time (years)	No. of effect sizes contributed by Raunkiaer subgroup	
Cole and Bayfield (1993)	Temperate coniferous forest, subalpine grassland	25, 75, 200, 500, 700 (Acute)	4	1	Chamaephyte	4
					Hemicryptophyte	4
					Geophyte	4
Cole (1995b)	Alpine, subalpine/montane grass/shrubland, temperate coniferous/ deciduous forest	10, 25, 50, 75, 100, 200, 250, 500 (Acute)	4	1	Chamaephyte	17
					Hemicryptophyte	40
					Geophyte	15
Cole and Spildie (1998)	Temperate coniferous forest	25, 150 (Acute)	4	1	Chamaephyte	2
					Geophyte	2
Monz et al. (2000)	Subalpine/montane grass/shrubland	25, 75, 200, 500, 800 (Acute)	4	1	Hemicryptophyte	16
Cole and Monz (2002)	Alpine, subalpine/montane grass/shrubland, temperate coniferous/ deciduous forest	75, 225, 600, 1500, 2400, 3000 (Chronic over 3 years)	4	3	Chamaephyte	8
					Hemicryptophyte	12
Monz (2002)	Alpine/tundra	25, 75, 200, 500 (Acute)	4	4	Chamaephyte	4
					Helophyte	4
Ikeda (2003)	Temperate grassland	80, 160, 640, 2560 (Chronic over 3.3 years)	3	3.3	Therophyte	4
Gallet et al. (2004)	Temperate shrubland	100, 200, 400, 800 (Acute, and Chronic over 0.2 years)	3	0.2	Phanerophyte	32
Roovers et al. (2004)	Temperate deciduous forest, temperate shrubland	25, 75, 250, 500 (Acute)	4	2	Phanerophyte	8
					Chamaephyte	12

Outcomes of multiple meta-regressions for Raunkiaer life-form subgroup analyses

Table 2. Outcomes of multiple meta-regressions for Raunkiaer life-form subgroup analyses

	Resistance			Recovery Time			Trampling intensity		
	Coeff.	z	*p*	Coeff.	z	*p*	Coeff.	z	*p*
Chamaephytes	-0.300	-3.69	<0.001	0.495	1.93	0.054	-0.001	-1.16	0.247
Hemicryptophytes	-0.233	-4.74	<0.001	-0.374	-1.95	0.051	0.001	0.97	0.330
Geophytes	-0.411	-3.41	0.001		N/A		0.002	1.26	0.206