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Biodiversity mediates the effects of stressors but not nutrients on litter decomposition

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- Abstract Understanding the consequences of ongoing biodiversity changes for ecosystems is a
 pressing challenge. Controlled biodiversity-ecosystem function experiments with random
 biodiversity loss scenarios have demonstrated that more diverse communities usually provide
 higher levels of ecosystem functioning. However, it is not clear if these results predict the
- ecosystem consequences of environmental changes that cause non-random alterations in
 biodiversity and community composition. We synthesized 69 independent studies reporting 660
- observations of the impacts of two pervasive drivers of global change (chemical stressors and
- ¹⁸ nutrient enrichment) on animal and microbial decomposer diversity and litter decomposition.
- ¹⁹ Using meta-analysis and structural equation modelling, we show that declines in decomposer
- ²⁰ diversity and abundance explain reduced litter decomposition in response to stressors but not to
- ²¹ nutrients. While chemical stressors generally reduced biodiversity and ecosystem functioning,
- 22 detrimental effects of nutrients occurred only at high levels of nutrient inputs. Thus, more intense
- ²³ environmental change does not always result in stronger responses, illustrating the complexity of
- ecosystem consequences of biodiversity change. Overall, these findings provide strong empirical
- ²⁵ evidence for significant real-world biodiversity-ecosystem functioning relationships when human
- activities decrease biodiversity. This highlights that the ecosystem consequences of observed
 biodiversity change are nontrivial and depend on the kind of environmental change.

29 Introduction

28

Human activities cause global environmental changes with important consequences for biodiversity 30 and the functioning of ecosystems. Understanding these consequences is crucial for better policy 31 and conservation strategies, which will ultimately promote human well-being too (IPBES, 2019). 32 A key question is to what extent changes in ecosystem functioning are mediated by changes at 33 which dimensions of biodiversity. Extensive research has demonstrated that biodiversity is needed 34 for the stable provenance and enhancement of ecosystem processes and functions (Cardinale 35 et al., 2012; Schuldt et al., 2018; Tilman et al., 2012). However, this body of evidence is mostly 36 based on experiments comparing ecosystem functioning in artificial communities with varying 37 number of species. Such experiments might not capture the complex ways by which shifts in 38 biodiversity induced by global change ultimately affect ecosystem functioning (De Laender et al., 39 2016; Eisenhauer et al., 2019b).

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Early biodiversity-ecosystem function (BEF) experiments typically controlled for environmen-41 tal gradients, thus not accounting for the underlying drivers of biodiversity change (De Laender 42 et al., 2016: Srivastava and Vellend, 2005: Wardle, 2016). These early experiments also focused on 43 species richness as the sole biodiversity index, and manipulated it directly and randomly. How-44 ever, environmental change will often elicit non-random changes in several facets of biodiversity 45 (Eisenhauer et al., 2016; Giling et al., 2019; van der Plas, 2019) (community composition and pop-46 ulation densities (Glassman et al. 2018: Spack et al. 2017) functional diversity (Cadotte et al. 47 2011: Craven et al., 2018: Heemsbergen et al., 2004), trophic diversity (Soliveres et al., 2016: Wang 48 and Brose, 2018; Zhao et al., 2019), or intra-specific diversity (Des Roches et al., 2018)). The se-49 lective effects of environmental change emerge because organisms differ in their response to 50 environmental change. For example, larger organisms and predators are often more negatively 51 affected than smaller organisms at lower trophic levels (Hines et al., 2015: Sheridan and Bickford, 52 2011: Srivastava and Vellend, 2005: Voigt et al., 2007). Using realistic extinction scenarios, exper-53 iments found contrasting effects of non-random shifts in biodiversity on ecosystem functioning 54 (e.g. Cárdenas et al., 2017: Jonsson et al., 2002: Melguizo-Ruiz et al., 2020: Oliveira et al., 2019: 55 Smith and Knapp, 2003; Zavaleta and Hulvey, 2004). In addition, several variables that are not 56 directly related to biodiversity control ecosystem functions (e.g. physiological rates (Dib et al., 57 2020: Thokur et al., 2018) and alterations of physical and chemical conditions (De Laender et al., 58 2016; Giling et al., 2019)). When environmental change affects these mechanisms, teasing out the 59 relative importance of biodiversity-mediated effects is complicated even more. Given the number 60 of different potential mechanisms, quantifying the extent to which shifts in biodiversity underpin 61 the effect of environmental change on ecosystem functioning under real-world scenarios of global 62 change is a key challenge for ecology (De Laender et al., 2016; Duffy et al., 2017; Eisenhauer et al., 63 2019b: Srivastava and Vellend, 2005: van der Plas, 2019: Wardle, 2016). Incorporating the impacts 64 of environmental change drivers into BEE studies and meta-analyses is an important step forward 65 to address such questions (De Laender et al., 2016: Eisenhauer et al., 2019b). 66 The vast majority of BEF experiments has focused on plant richness and ecosystem functions 67 such as biomass production (van der Plas. 2019). However, litter decomposition has a tremendous 68 importance in ecosystems and biogeochemical cycles (Follstad Shah et al., 2017). Small changes in the rate of this process can have important consequences for the overall carbon balance. Indeed, 70 increases in decomposition rates could have positive feedback effects on climate warming by 71 enhancing C losses (Kirschbaum, 2000). The diversity of decomposers (invertebrates and micro-72 organisms that fragment and decompose organic matter in both aquatic and terrestrial systems) 73 is crucial for litter decomposition (Eisenhauer et al., 2012: García-Palacios et al., 2013: Gessner 74 et al., 2010: Handa et al., 2014: Hättenschwiler et al., 2005) and for other ecosystem functions as 75 well (Eisenhauer et al., 2019a: Lefcheck et al., 2015: Schuldt et al., 2018). Despite the importance 76 of decomposers. BEF experiments focusing on litter decomposition more often addressed the 77 influence of plant litter diversity than of decomposers (Gessner et al., 2010; Tonin et al., 2018). In 78 a meta-analysis, decomposer diversity had a greater effect on decomposition than the diversity 79 of plant litter (Srivastava et al., 2009), although also weak and neutral effects have been reported 80 (van der Plas, 2019). Facilitation and complementarity through niche partitioning are primary 81 mechanisms underlying the positive relationship between decomposer diversity and decomposition 82 (Gessner et al., 2010; Hättenschwiler et al., 2005; Tonin et al., 2018). Experiments conducted 83 in natural conditions and reflecting realistic extinction scenarios are still relatively scarce, and 84 demonstrate contrasting effects of non-random shifts in decomposer diversity on decomposition 85 (Cárdenas et al., 2017: Jonsson et al., 2002: Melguizo-Ruiz et al., 2020). The need to quantify 86 environmental change effects on decomposer diversity, along with potential knock-on effects on 87 litter decomposition, is therefore particularly pressing. 88 There is a variety of environmental change drivers, and different types of drivers may have 89 diverse effects on biodiversity and ecosystem functions (De Laender et al., 2016: Dib et al., 2020).

¹ We postulate that there are two main categories of environmental change: stressors and resource

shifts. While stressors cannot be consumed, and act as conditions that alter growth rates (e.g., temperature, drought, chemical stressors), resources are by definition consumed (e.g., CO_2 or mineral 93 nutrients), which has important implications for how they should enter theory (Chase and Leibold, 94 2003: De Loender, 2018). Chemical stressors and nutrient enrichment are important case studies 95 of environmental stressors and resource enrichment because their presence is increasing rapidly (Bernhardt et al., 2017) and they are projected to have severe effects on biodiversity (Mazor et al., 97 2018) They are also of particular relevance for decomposer communities. Chemical stressors such 98 as metals and pesticides decrease the diversity, abundance, growth and activity of decomposers ac across terrestrial and aquatic systems (e.g., Hogsden and Harding, 2012: Pelosi et al., 2014: Schäfer, 100 2019). In contrast, nutrient enrichment can have positive impacts on the abundance and physio-101 logical rates of decomposer organisms by reducing resource limitations (*Treseder, 2008*), but at 102 the same time decrease decomposer diversity (Lecerf and Chauvet, 2008: Woodward et al., 2012). 103 Across ecosystems, stressors and nutrients can exert opposite impacts on litter decomposition 104 rates, with decreases in response to chemical stressors but increases following nutrient enrichment 105 (Ferreira et al., 2015, 2016). In addition, decomposition involves both microorganisms and inverte-106 brates (Bardgett and van der Putten, 2014: Gessner et al., 2010: Hättenschwiler et al., 2005) that 107 may respond differently to stressors and nutrients with a higher sensitivity of invertebrates than 108 microorganisms (Peters et al., 2013: Siebert et al., 2019). Although many published case studies 109 report shifts in decomposer diversity and in rates of litter decomposition at sites impacted by 110 stressors and nutrients, biodiversity-mediated effects have not yet been quantified across systems. 111 Here we addressed the question if the effects of stressors and nutrient enrichment on decom-112 poser diversity and abundance explain the response of litter decomposition to these two types 113 of pervasive environmental change drivers (*Figure 1*). We synthesized 69 published case studies 114 reporting the impact of stressors (metals, pesticides) and nutrients (nitrogen or phosphorous 115 additions) on litter decomposition and on decomposer diversity (taxa richness, Shannon diversity, 116 evenness) or abundance (density, biomass) at sites differing in stressor or nutrient levels. Our 113 comprehensive global dataset of 660 observations encompasses studies across taxonomic groups 118 (animal (soil micro-, meso- and macrofauna, stream macroinvertebrates) and microbial (fungi and 119 bacteria) decomposers), ecosystems (aquatic and terrestrial), and study types (experimental and 120 observational) (Figure 2). We quantified the effect size of environmental change on decomposer 121 diversity or abundance and on litter decomposition within studies using correlation coefficients 122 between stressor or nutrient levels and decomposer diversity, abundance, and litter decomposition. 123 We also characterized stressor and nutrient intensities, and standardized their levels in water, soil 124 or sediment using environmental quality criteria issued by environmental authorities (e.g. ECHA. 125 USEPA, UKTAG). Using meta-analysis and structural equation modelling (SEM), we first compared 126 the overall effects of stressors and nutrients on decomposers and decomposition across systems 127 and studies (first meta-analysis), and second, addressed to what extent changes in decomposer 128 diversity and abundance mediate the impacts of these two contrasting drivers of environmental 129 change on decomposition (second meta-analysis and SEM). Third, we explored the effects of three 130 main moderators on decomposers diversity, abundance, and decomposition responses, as found 131 in the second meta-analysis: stressor or nutrient intensity, taxonomic group (animal vs. microbes) 132 and study type (experimental vs. observational studies). 133 We expected that chemical stressors and nutrients would have contrasting effects on de-134 135

We expected that chemical stressors and nutrients would have contrasting effects on de composer diversity and abundance, and on litter decomposition across ecosystems and studies
 (*Figure 1*). We hypothesized that chemical stressors generally decrease decomposer diversity,
 abundance (*Hogsden and Harding, 2012; Petrin et al., 2008*), and litter decomposition rates (*Fer <i>reira et al., 2016; Peters et al., 2013*), and that nutrients generally decrease decomposer diversity
 (*Lecerf and Chauvet, 2008; Woodward et al., 2012*) but increase decomposer abundance and litter
 decomposition rates (based on physiological effects and decreasing resource limitations (*Bergfur et al., 2007; Ferreira et al., 2015; Treseder, 2008; Woodward et al., 2012*)). We further hypothesized
 that litter decomposition responses to environmental change depend on changes in decomposer di-



Figure 1. Schematic representation of the structural hypotheses tested in this study. Green arrows depict expected positive effects, red arrows represent negative effects. Stressors and nutrients are hypothesized to decrease decomposer diversity. The response of decomposers diversity to environmental change drivers determines the response of decomposition (Srivastava et al., 2009). Nutrients are hypothesized to increase decomposer abundance. Stressors and nutrients can affect litter decomposition independent of changes in decomposer diversity and abundance, especially through changes in physiological activity (De Laender et al., 2016, Giling et al., 2019).

versity and abundance, and expected an overall positive relationship independent of environmental
 change intensity (*Srivastava et al., 2009*).

145 **Results**

146 Description of the data and overall patterns

The final dataset contained 69 (case) studies from 59 publications, representing 660 observations. 147 Data were mostly from Europe (44; 443 (studies; observations)) and North and South America (19; 148 168), while Asia (2; 9) and Oceania (4, 40) were less well represented (*Figure 2*.A). The studies covered 149 aquatic (55; 388) and terrestrial systems (14; 272) (Figure 2.C), and used observational (43; 336) or 150 experimental approaches (26; 324). Studies reported abundance (66; 463) or diversity responses 151 (48; 197) (Figure 2.B) of soil and benthic invertebrates (48; 509) and microbes (fungi and bacteria) 152 associated with litter materials (36: 151) (Figure 2.C). Chemical stressors were mostly metals (13: 153 257) and pesticides (12; 66) associated with industrial activities, accidental spills, and agricultural 154 practices. Nutrient enrichment studies addressed fertilization by various N and/or P forms (26: 155 175), and eutrophication due to agricultural runoffs (10; 59) or wastewater effluents (4; 44). There 156 was no study reporting nutrient enrichment impacts on soil decomposer diversity in the dataset. 157 Funnel plots and intercepts of Egger's regression showed evidence for positive publication bias in 158 nutrient enrichment studies reporting decomposer abundance (Appendix 2-Figure 2, -Table 1). No 159 publication bias was detected in the other datasets. 160 We found largely contrasting effects of stressors and nutrients on each of the three response 161 variables in a first-level meta-analysis comparing the overall effects of the two drivers of environ-162 mental change (Figure 3, Appendix 2-Table 2). Chemical stressors overall decreased decomposer 163 diversity, abundance and litter decomposition across studies (Figure 3). Nutrient enrichment 164 tended to decrease decomposer diversity but to increase abundance, and decomposition, although 165 these trends were not significant as indicated by confidence intervals of the grand mean effects 166

¹⁶⁷ overlapping with zero (*Figure 3*).

Biodiversity-mediated effects of stressors and nutrients on litter decomposition

The responses of decomposition and of decomposer diversity and abundance to chemical stressors were correlated: decreases in decomposition were associated with decreases in decomposer diversity and abundance (*Figure 4* upper panels). We did not find such a relationship for nutrients.





Figure 2. Description of the data used in the present meta-analysis. A: countries represented and corresponding number of observations, B: decomposer diversity and abundance metrics covered, and C: ecosystem types and decomposer taxonomic groups (animals: soil micro-, meso-, macro-fauna, stream macroinvertebrates; and microbial decomposers: fungi and bacteria) represented.



Figure 3. Grand mean effect sizes of chemical stressors and nutrient enrichment on decomposer diversity (taxa richness and diversity indices), abundance (density and biomass), and litter decomposition. Effect sizes are z-transformed correlation coefficients. Error bars show 95% confidence intervals. Numbers in parentheses indicate number of studies and observations, respectively. Symbols show the significance level for the comparison between mean effect size and zero (*** *P* <0.001; * *P* <0.05). For full model results, see *Appendix 2-Table 2*.

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Figure 4. Relationship between the responses of decomposition and decomposer diversity and abundance to chemical stressors and nutrient enrichment. Variables are effect sizes (z-transformed correlation coefficients) of stressors or nutrients on litter decomposition and on animal and microbial decomposer diversity (left panels) or abundance/biomass (right panels). Gray symbols are individual observations of effect sizes; Colored symbols indicate the mean effect size on biodiversity or abundance across effect sizes on litter decomposition. Darker colors represent a higher standardized level of environmental change. Lines represent meta-regressions between effect sizes for decomposition and decomposers, where solid lines are statistically significant (P < 0.05), dashed lines are non-significant (P > 0.05), and thin lines depict the regressions, respectively, with sample size (number of studies; number of observations).

Instead, a range of positive and negative responses of decomposer diversity, abundance, and 172 decomposition to nutrients were found, without significant associations between them (Figure 4 173 lower panels). In addition, when decomposer diversity and abundance responses to nutrients were 174 close to zero, there was a wide range of decomposition responses (intercepts from *Figure 4* lower 175 panels). 176 According to our overarching hypothesis, the SEM indicated that the effects of stressors on litter 177 decomposition were mediated by shifts in decomposer diversity and abundance. Including the 178 direct paths from decomposer diversity or abundance to litter decomposition improved both the 179 models according to mediation tests and AIC comparisons (Figure 5). In addition, the path coeffi-180 cients from diversity and abundance to the decomposition response to stressors had (standardized) 181

values higher than 0.1 (*Figure 5*) and were statistically different from zero (*Appendix 2-Table 3*).
 However, in contrast to chemical stressors, the SEM did not support biodiversity-mediated effects
 of nutrient enrichment on litter decomposition. While the mediation test and AIC indicated that
 the decomposer diversity-mediated path improved the model (*Figure 5*), the path coefficient was

not significantly different from 0 (*Appendix 2-Table 3*). The decomposer abundance-mediated path

of nutrients was not supported by the data: an SEM without the direct path from decomposer

abundance to decomposition could not be rejected based on the mediation test (*Figure 5*), and including this path did not improve the model according to the AIC comparison. Besides, we found publication bias in this dataset (*Appendix 2-Figure 2*, *-Table 1*), and model check indicated that the residuals of the nutrients-abundance model were non-independent from the fitted values. Thus, the results from this model are reported here for comparison purposes only.

The magnitude of the biodiversity-mediated effects of chemical stressors on decomposition was 193 stronger than that of the direct effects of stressor intensity on decomposition. The indirect effect of 194 stressors on decomposition mediated by diversity (i.e. mathematical product of the standardized 195 paths from stressor intensity to decomposer diversity and from diversity to decomposition Figure 5) 196 was higher than the direct effect of stressors on decomposition, while the abundance-mediated 197 effect of stressors was negligible (*Figure 5*). In the case of nutrient enrichment, however, decomposi-198 tion responses were not explained by shifts in decomposer diversity and abundance, and the direct 190 effects of nutrient intensity dominated the total effect (Figure 5). Finally, between-model compar-200 isons (based on unstandardized path coefficients (Grace, 2006)) revealed that decomposer diversity 201 was a stronger driver of decomposition response to stressors than decomposer abundance (unstan-202 dardized paths were 0.42 and 0.24 respectively for diversity and abundance. Appendix 2-Table 3). 203 Sensitivity analyses revealed that the results were robust to the inclusion of approximated 204

standard deviations (*Appendix 3- -Table 1, -Table 2*), and extreme values of effect sizes (*Appendix 3-Table 3, -Table 4*). We found partially different results when using log-response ratios as effect
 sizes (*Appendix 3-Table 5, -Table 6*), due to lower sample sizes and emergence of extreme values in
 these datasets. In addition, the log-response ratio is probably sensitive to the various metrics of
 biodiversity, abundance, and decomposition covered by the individual studies that we included,
 while correlation coefficients better accommodate such discrepancies (*Koricheva et al., 2013*).

Response of animal and microbial decomposers and decomposition to stressor and nutrient intensity

Despite the overall negative effects of stressors on decomposition, negative responses in decompo-213 sition were not associated with higher stressor intensity (Figure 5, Figure 6). This result held for two 214 complementary approaches: multivariate SEM (Figure 5) that relied on data resampling to account 215 for replicated values of decomposition matching several decomposer responses (e.g. for different 216 taxa in the same litterbag), and meta-regressions (Figure 6) where data resampling was not neces-217 sary (see Methods). There was mixed support for a stressor intensity effect on decomposer diversity 218 across the two approaches: decomposer diversity responses decreased with stressor intensity 219 according to the SEM (*Figure 5*), but this trend was not significant according to the second level 220 meta-analysis (*Figure 6*). Similar slopes were obtained both with the SEM relying on data resampling 221 (the slope of the relationship was -0.10 + 0.04 Appendix 2-Table 2) and with the meta-regression 222 (the slope was -0.05 ± 0.03). The differences between the two approaches can be explained by 223 the different data included. Decomposer abundance responses were not associated to stressor 224 intensity in both the SEM and meta-regression approaches (Figure 5, Figure 6). We found different 225 patterns for nutrient enrichment, where decomposition responses decreased with nutrient intensity 226 (Figure 5, Figure 6), from positive effects at low intensity to negative effects at higher intensity (Fig-227 *ure 6*). A similar pattern was observed for decomposer diversity, where responses decreased with 228 nutrient intensity from positive to neutral to negative responses at high nutrient levels (Figure 6) 220 Nutrient intensity, however, did not explain the responses of decomposer abundance (Figure 5. 230 *Figure 6*), and both positive and negative responses were found at high nutrient levels. 231 The meta-analysis further revealed clear discrepancies between the response of animal and 232

microbial (fungi and bacteria) decomposers to stressors and nutrients. Animal decomposers responded more strongly to chemical stressors than microbial decomposers. The mean effects of chemical stressors on animal decomposer diversity and abundance were more negative than that on microbial decomposers, confirmed by Wald type tests of the second-level meta-analyses (*Figure 7* upper panels, *Appendix 2-Table 4*). Animal decomposers overall decreased in diversity but



Figure 5. Decomposer diversity and abundance explained litter decomposition response to chemical stressors but not to nutrient enrichment. Structural equation models investigating decomposer diversityor abundance-mediated effects of chemical stressors and nutrient enrichment on litter decomposition across 69 studies. Arrows represent relationships between stressor or nutrient intensity levels, and effect sizes of stressors or nutrients on litter decomposition and on decomposer diversity (taxa richness, Shannon diversity, or evenness: left panels) or abundance and biomass (right panels). Values along the arrows are standardized path coefficients. Green, red, and gray arrows indicate positive, negative, and non-significant relationships, respectively. Curved arrows depict the indirect effects of stressors or nutrients on decomposition as mediated by diversity or abundance. Arrow widths are scaled relative to the magnitude of standardized path coefficients. C statistic, *P*-value (*P* <0.05 indicate poor model fit), and sample sizes (number of studies; number of observations). Results of mediation tests: comparison with models omitting the path from diversity or abundance to decomposition (Δ AlC < -2 indicates that reduced models were not consistent with the data).



Figure 6. Decomposer and decomposition responses to the intensity levels of chemical stressors and nutrient enrichment. Values are effect sizes (z-transformed correlation coefficients). Stressor or nutrient intensity represents the standardized level of environmental change in the treatment with the highest level (values <0: observed level below quality criteria considered to be safe for the environment; values > 0: observed level above quality criteria). Point size is proportional to the inverse of the variance in effect size. Lines are the slopes and 95% confidence intervals from bivariate meta-regressions, with associated QM statistics, *P*-value and sample size (number of studies; number of observations).





increased in abundance in response to nutrient enrichment (*Figure 7*, lower panels). On the other
 hand, the mean effects of nutrients on microbial decomposer diversity and abundance had lower
 magnitudes compared to animals (*Appendix 2-Table 4*), with confidence intervals overlapping with
 zero (*Figure 7* lower left panel). Finally, there was no clear difference between observational and
 experimental studies (*Figure 7, Appendix 2-Table 4*), and between biodiversity responses in terms
 of taxa richness or of diversity indices (*Appendix 2-Table 4*).

244 **Discussion**

The present synthesis brings new insights into how changes in decomposer biodiversity induced 245 by two pervasive drivers of environmental change ultimately affect decomposition. We find con-246 comitant changes in biodiversity and decomposition under the influence of chemical stressors 247 but not nutrient enrichment, highlighting that real-world patterns relating shifts in biodiversity 248 and ecosystem functioning depend on the type of environmental change. In fact, we observed 249 significant correlations between effects on biodiversity and ecosystem function in a scenario where 250 chemical stressors caused a significant decline in biodiversity. In contrast, in cases where nutri-251 ent enrichment caused variable responses in biodiversity, relationships between biodiversity and 252 ecosystem function responses were weaker. It remains an understudied but important question if 253 results of controlled BEF experiments are applicable to non-random changes in biodiversity caused 254 by human activities (e.g., De Laender et al., 2016; Duffy et al., 2017; Eisenhauer et al., 2019b; Sri-255

vastava and Vellend, 2005; van der Plas, 2019; Wardle, 2016). The present results provide strong

²⁵⁷ empirical evidence for significant real-world BEF relationships when human activities decrease

258 biodiversity.

259 Biodiversity-mediated effects of chemical stressors on decomposition

Chemical stressors caused consistent reductions in decomposer diversity and abundance as well as 260 in litter decomposition rates in line with several previous case studies (*Beketov et al. 2013: Malai* 261 et al., 2014) and meta-analyses (Ferreira et al., 2016: Peters et al., 2013). Adding to the previous 262 knowledge, the present meta-analysis shows that changes in decomposer diversity and abundance 263 explained the decomposition response to stressors, providing evidence for the expectation that 264 shifts in biodiversity mediate the impact of chemical stressors on decomposition. We acknowledge 265 that despite the SEM analysis, the approach conducted here remains correlative. However, our 266 study builds on a body of experimental and observational evidence that already demonstrated that 267 more diverse and abundant decomposer communities support higher decomposition rates, albeit 268 not under the influence of environmental change (e.g. García-Palacios et al., 2013; Handa et al., 269 2014). 270

We especially complement a previous meta-analysis showing the importance of decomposer 271 diversity for decomposition across experiments manipulating the richness of invertebrate and 272 microbial decomposer communities (*Srivastava et al., 2009*). We extend on this and show that 273 non-random biodiversity losses induced by stressors are closely associated with decreases in 274 decomposition across a wide range of studies. A recent review pointed out that in naturally-275 assembled terrestrial communities, studies more often found neutral and to a lesser extent positive 276 relationships between decomposer diversity and decomposition (van der Plas. 2019). In that 277 review, communities were not influenced by environmental change drivers, and the vote counting 278 approach used is sensitive to the statistical power of individual studies and could have increased 279 the probability of finding non-significant relationships (Koricheva et al., 2013). In line with our 280 findings, an experiment mimicking the sequence in which freshwater invertebrate decomposers 28 are lost after disturbances showed that decreasing non-randomly the number of species decreased 282 decomposition rates (Ionsson et al., 2002). 283

Biodiversity-ecosystem function experiments manipulating biodiversity directly are key to un-284 derstand the mechanisms involved in this relationship (*Eisenhauer et al.*, 2016), especially because 285 they control for the effects of environmental beterogeneity or abundance. However, in real-world 286 scenarios environmental change drivers affect both biodiversity and abundance simultaneously. As 287 demonstrated here, this is especially the case for stressors that decrease decomposer diversity and 288 abundance (Hogsden and Harding, 2012). The abundance or biomass of different decomposers 280 is of critical importance for decomposition (e.g. Bergfur et al., 2007; Ebeling et al., 2014; Manning 290 and Cutler, 2018). Even at constant richness and community composition, strong decreases in 291 abundance can have important impacts on ecosystem functioning (Spagk et al., 2017: but see 292 Dainese et al., 2019). It is beyond the scope of the present meta-analysis to disentangle the effects 293 of biodiversity from the effects of abundance, and we found that both contributed to explain shifts 294 in decomposition in separate analyses. It is interesting to note that the few cases where negative 205 effect sizes of stressors on biodiversity were associated with positive effect sizes on decomposition 296 were also cases where decomposer abundance was positively associated with stressors (Figure 4) 297 Although we cannot specifically test this with the present data, it seems that in those particular 298 cases, increases in decomposer abundance counteracted the negative effects of decreases in 299 decomposer diversity (*Lucising et al.*, 2015). Those results could therefore be in line with the mass-300 ratio hypothesis (Grime, 1998; Smith and Knapp, 2003). Indeed, an exclusion experiment showed 301 that dominant, small, detritivores can compensate reductions in litter decomposition caused by 302 the removal of large detritivores (*Cárdenas et al., 2017*). These concomitant shifts in both diversity 303 and abundance further have important implications for our estimates of diversity responses, as studies mostly reported richness to estimate decomposer diversity, but rarely corrected for the 305

sampling effort (*Gotelli and Colwell, 2001*). This means that lower abundances rather than a lower
 number of species *per se* might have directly caused some of the negative effects on biodiversity
 reported here (*Chase and Knight, 2013*). This common caveat in meta-analysis approaches that
 rely on how individual studies report biodiversity, also applies to the present study, and reinforces
 the importance of reporting raw data in future studies of the impacts of chemical stressors on
 biodiversity and ecosystem functioning.

The effects of changes in decomposer diversity and abundance on decomposition found in 312 the present study might also have channeled changes in community and food-web structure 313 not captured by our biodiversity metrics. Changes in keystone species (Hättenschwiler et al., 314 2005), functional diversity (Cadotte et al., 2011; Dangles et al., 2012; Heemsbergen et al., 2004), 315 vertical diversity (Gessner et al., 2010: Melguizo-Ruiz et al., 2020: Wang and Brose, 2018: Zhao et al., 316 2019), or dominance patterns (Dangles and Malmavist, 2004) might have shifted concomitantly to 317 taxonomic diversity and abundance. Moreover, these different components of diversity might act 318 at different timings of decomposition (Oliveira et al., 2019). Unfortunately, studies rarely reported 319 such measurements together with decomposition. For example in our dataset, only 7 studies 320 reported evenness. Future studies need to explore shifts in decomposer community composition in 321 more detail to better understand what particular aspect of biodiversity is responsible for changes 322 in decomposition rates (Giling et al., 2019: Höttenschwiler et al., 2005). In particular, few of the 323 included studies reported comparable functional groups allowing to address the effect of functional 324 diversity across the multiple systems and taxonomic groups addressed by the present analysis. 325 Future synthesis work could specifically address the effect of functional diversity, by focusing on a 326 given system type. Indeed, there is ample evidence that shifts in functional diversity are crucial for 327 decomposition (*Heemsbergen et al.*, 2004), and that facilitative interactions occur primarily between 328 decomposers of contrasting body size (Dangles et al., 2012; Tonin et al., 2018). This is especially the 329 case for interactions between animal and microbial decomposers, where fragmentation of litter by 330 detritivores facilitates access for microbial decomposers (Eisenhauer et al., 2010: Hättenschwiler 33. et al., 2005: Yang et al., 2012). 332

Here, we found that invertebrates were more affected by chemical stressors than microbes. 333 across aquatic and terrestrial ecosystems. Invertebrate decomposers are particularly sensitive to 334 the impacts of metals and pesticides (Hogsden and Harding, 2012: Pelosi et al., 2014: Peters et al., 335 2013: Schäfer, 2019). Microbial decomposers are known to be sensitive to metals (Giller et al. 336 2009) and pesticides as well (DeLorenzo et al., 2001). Nevertheless, our result is consistent with the 337 general expectation that larger organisms are more sensitive to environmental change due to longer 338 generation time, higher energetic demands and lower population densities (*Hines et al., 2015*: 339 Sheridan and Bickford, 2011: Woodward et al., 2005: Yvon-Durocher et al., 2011). These different 340 sensitivities between groups of decomposers could imply that the biodiversity-mediated effects of 3/1 stressors on decomposition are more strongly linked to shifts in invertebrates than microbes, as 342 reported in a previous review (*Peters et al.*, 2013). However, in another meta-analysis focusing on 343 microbial-driven decomposition rates, changes in fungal biomass and richness explained shifts in 344 decomposition under the impacts of chemical stressors, but also of nutrient enrichment (Lecerf 345 and Chauvet, 2008). 346

Nutrient-induced changes in decomposition were not related to shifts in decom poser diversity

The impacts of nutrient enrichment on litter decomposition and decomposer diversity were different from those caused by stressors, confirming our expectations. These different biodiversity and function responses led to different emergent relationships between decomposer diversity and decomposition compared to stressors. We found that nutrients had a variety of effects ranging from positive to negative depending on the taxonomic group (*Figure 7*) and nutrient intensity (*Figure 6*), and resulting in neutral overall mean effects (*Figure 3*). Previous syntheses also found positive (*Ferreira et al., 2015*) as well as inconsistent (*Knorr et al., 2005*) responses of decomposition rates

to nutrient enrichment in streams. The relatively small mean effect of nutrient enrichment on 356 decomposition in the present meta-analysis could be explained by the use of correlation as an effect 357 size, which does not capture potentially non-monotonic responses of decomposition to nutrients 358 (Woodward et al., 2012). However, we noted that most of the studies included in the present 359 meta-analysis did not individually span nutrient gradients sufficiently large to capture this potential 360 non-monotonous response. Taken together, the studies show positive effects on decomposition 361 at low nutrient intensities that shifted towards neutral to negative effects at higher intensities 362 (Figure 6), which is consistent with previous findings (Ferreira et al., 2015; Woodward et al., 2012) 363 Low nutrient intensities might have enhanced microbial activity and biomass by alleviating resource 364 limitation, resulting in enhanced decomposition. At higher intensities, however, negative impacts 365 on invertebrates might have decreased decomposition rates (Peters et al., 2013; Woodward et al., 366 2012) 367

These nutrient intensity patterns contrasted with the results for chemical stressors. The overall 368 negative effects of stressors (Figure 1) on decomposition were not explained by stressor intensity 369 levels (*Figure 6*), and there was mixed support for a stressor intensity effect on decomposer diversity 370 based on two complementary data analysis approaches (SEM based on data resampling (Figure 5) 371 vs. second level meta-analysis *Figure 6*). Thus, negative responses to chemical stressors happened 372 across the range of stressor intensity. Such contrasting patterns between stressor and nutrient 373 intensity effects may reflect the greater number of stressor types (different metals, pesticides, 374 mixtures) covered by individual studies compared to the limited number of nutrients. In addition, 375 due to the higher variability of stressor types, we relied on more variable sources to standardize 376 stressor levels compared to nutrients in the diversity dataset (Methods, Appendix 1-Table 1). With 37 the data at hand, it was not possible to test the influence of the environmental quality criteria 378 used to standardize stressor and nutrient levels, because such an effect would be confounded with 379 stressor or nutrient types. The datasets were all dominated by environmental quality criteria based 380 on similar methodologies (for 75 to 100% of observations, see Methods). However, future studies 38 focusing on stressor intensity effects across ecosystems would greatly benefit from coordinated 382 efforts to derive quality criteria encompassing the vast and rapidly increasing number of chemical 383 stressors (Wang et al., 2020). 384

Contrary to our expectation, nutrient-induced shifts in decomposer diversity and abundance 385 were not associated with shifts in decomposition rates across studies. We found that increasing 386 nutrient intensity decreased the effects on decomposition and on decomposer diversity, but not 387 on decomposer abundance. Statistically controlling for the effect of nutrient intensity with SEM 388 indicated no residual association between shifts in decomposer diversity or abundance and in 380 decomposition rates, i.e. a non-significant BEF relationship. Changes in microbial abundance 390 in response to nitrogen deposition explained the responses of different ecosystem functions in 391 terrestrial systems in previous meta-analyses (García-Palacios et al., 2015: Treseder, 2008). Here 392 we show that this pattern cannot be generalized across aquatic and terrestrial systems and across 393 animal and microbial decomposers. Contrary to stressors, when the diversity and abundance of 394 animal and microbial decomposers were not affected by nutrients, we observed large positive 395 and negative shifts in decomposition (intercepts of *Figure 4*), that were explained by nutrient 396 intensity (Figure 4: negative effects on decomposition at invariant biodiversity are associated 397 with high intensities and positive effects with lower intensities). Together, these results show 398 that nutrient-induced shifts in decomposer diversity were not as strong drivers of decomposition 390 changes as stressor-induced biodiversity shifts. These differences may be partly due to the different 400 mechanisms underlying the effects of stressors and nutrients. Based on previous studies, we 401 speculate that our results are due to the complex responses of animal and microbial decomposers 402 at different nutrient intensities (Ferreira et al., 2015; Lecerf and Chauvet, 2008; Treseder, 2008; 403 Woodward et al. 2012) 404

Animal decomposers showed a stronger response to nutrients than microbes. Invertebrate decomposers overall decreased in diversity, but they increased in abundance under nutrient

enrichment. These results could reflect a loss of sensitive taxa to the benefit of tolerant taxa that 407 were able to use additional resources and would then increase in density (*Bergfur et al., 2007*). 408 Overall, microbial decomposers responded little to nutrient enrichment, probably reflecting a 409 mixture of positive and negative effects that nutrients can have on microbial growth (Lecerf and 410 Chauvet 2008: Treseder 2008) as well as on different microbial taxa. Indeed nutrients can alleviate resource limitations at low intensities, but can also exert toxic effects at high intensities. The initial 412 levels of nutrients thus condition subsequent responses in decomposers and decomposition to 413 nutrient enrichment (Ferreira et al., 2015: Knorr et al., 2005), Furthermore, at high intensities. 414 nutrients can be associated with other chemical stressors (e.g. pesticides in agricultural runoffs) 415 (Ferreira et al., 2015: Woodward et al., 2012). The influence of interactive effects of stressors and 416 nutrients was impossible to quantify with the data at hand, given that only a few experiments 417 assessed the effects of both drivers independently, but many observational studies may have 418 been confounded by such joint effects. Chemical stressors and nutrients are often co-occurring in 419 e.g. agricultural landscapes, and the consequences of such combinations are still poorly understood. 420 Furthermore, stressor and nutrient effects might be modulated by climatic and other environmental 421 conditions, and studies on interaction effects are scarce (Rillig et al., 2019; Thakur et al., 2018) 422 Finally, although our comparison of stressors versus resources allowed us to test a clear concept. 423 any kind of grouping in ecological studies may mask some of the variation within the categories 424 and future studies may be interested in different categories. As data availability improves, future 425 work could include different environmental change drivers. This would also allow to test additional 426 groupings of drivers and ecological concepts unifying stressors and resources (De Laender, 2018; 427 Harlev et al., 2017). 428

429 Conclusions

In conclusion, this study brings new insights into the real-world patterns relating ecosystem function 430 to non-random changes in biodiversity induced by environmental change. We found that the conse-431 guences of changes in biodiversity for ecosystem functioning depend on the type of environmental 432 change. Real-world scenarios do not necessarily involve concomitant changes in both biodiversity 433 and function across terrestrial and aquatic systems. We further found that with the environmental 434 guality criteria used in risk assessment, there were already significant positive and negative effects 435 on decomposers and decomposition (*Figure 6*), highlighting the need to better incorporate biodi-436 versity and ecosystem function into ecological risk assessment programs (De Laender and Janssen) 437 2013). Finally, we report overall negative effects of chemical stressors on biodiversity and ecosys-438 tem functioning across terrestrial and aquatic ecosystems that reinforce recent calls to consider 430 chemical stressors as important global change drivers and address their impacts on biodiversity 440 and ecosystems (Bernhardt et al., 2017: Mazor et al., 2018: Steffen et al., 2015). Positive real-world 441 BEF relationships may be particularly significant in cases where environmental changes decrease 112 biodiversity, such as in the case of chemical stressors. Such information are crucial if we are to 443 design policy and conservation strategies able to reconcile human development with biodiversity 444 conservation. 115

446 Methods

447 Data collection

We searched the Web of Science for studies that addressed the impact of environmental drivers and
recorded decomposer community responses and litter decomposition rates. The search strategy
is fully reported in Supplementary Methods (*Appendix 1*). The search retrieved 2536 references.
Abstracts and titles were screened to identify a final set of 61 records that met our inclusion criteria
(PRISMA plot, *Appendix 1-Figure 1*, and list of included references (*Appendix 4*)). To be included in
the meta-analysis, studies had to:

• Report litter decomposition (rates, mass loss, proportion of mass remaining) and the diversity,

- abundance, or biomass of decomposers at sites differing in chemical stressor or nutrient
 levels.
 - Focus on naturally-assembled communities subjected to the impact of chemical stressors or nutrient enrichment. Studies that manipulated decomposer diversity directly were not consid-
- nutrient enrichment. Studies that manipulated decomposer diversity directly were not consid ered to only focus on non-random biodiversity change scenarios. We included mesocosm
 studies only when they used field-sampled communities and left time for the community
 to reach an equilibrium in mesocosms in order to reflect real-world conditions as much as
- 462 possible.

457

- Report the response of animal (benthic macroinvertebrates, or soil micro, meso or macro fauna) or microbial decomposers (bacteria or fungi from decomposing leaves or in surround ing water or soil samples).
- Report decomposer abundance (density or biomass), or decomposer diversity (taxa richness,
 Shannon diversity, evenness).

When a reference reported different environmental change drivers or geographical areas with 468 a specific reference site for each case, we considered these as individual (case) studies (*García*-469 Palacios et al., 2015). We extracted means or sums, standard deviations, and sample sizes of litter 470 decomposition, decomposer diversity, and abundance (outcomes) in non-impacted vs. impacted 471 sites (control-treatment studies), or at each site when gradients of chemical stressors or nutrients 472 were investigated (gradient studies). When response variables were reported at different time 473 points, we kept only the last time point to capture long-term responses. For studies reporting 474 decomposition, decomposer abundance or diversity for several litter types (e.g. different litter 475 species), several groups of organisms (e.g. functional feeding groups for macroinvertebrates). 476 and several diversity metrics (e.g. Shannon indices and taxon richness), we created separate 477 observations within case studies. We also extracted chemical stressor or nutrient levels at those 478 sites (water, soil, or sediment concentrations of chemical stressors or nutrients, or application rate 470 of pesticides or fertilizers). The study type (experimental vs. observational), taxonomic group (animal 480 decomposers or microbial decomposers) and metric of diversity (taxa richness or diversity indices 481 (Shannon diversity and evenness)) were also recorded. We used the online software Webplotdigitizer 482 to extract data from figures (Rohatzi, 2018). We converted standard errors and confidence intervals 483 into standard deviations using the equations in Lajeunesse (2013). When reported as mass loss, 484 litter decomposition data were transformed into k rates using the exponential decay equation used 485 in Ferreira et al. (2015). 486

487 Effect size calculation

We used z-transformed correlation coefficients as effect sizes in order to cope with the heterogeneity 488 of data and study types (Koricheva et al., 2013). For control-treatment studies, we first calculated 489 Hedge's d, and then transformed Hedge's d into correlation coefficients (*Laieunesse, 2013*). For 490 gradient studies (4 or more treatment levels), we calculated correlation coefficients between the 491 mean values of abundance, diversity, or decomposition rate and the corresponding chemical 492 stressor or nutrient concentrations. When means, standard deviations, or sample sizes were 493 missing, we contacted the authors to retrieve the data. When the information could not be 494 retrieved, standard deviations were approximated from the data, using the linear relationship 495 between mean values and standard deviations across our datasets (Laieunesse, 2013). 496

497 Standardization of chemical stressors and nutrient enrichment intensities

Given the variability in the different stressors and nutrients combinations in the studies, stressor and
 nutrient levels were standardized into a common environmental change driver intensity (*ECD*_{intensity})
 as follows:

501 $ECD_{intensity} = log([Compound_i]_{treatment}/[Compound_i]_{criteria})$

where [*Compound*_i]_{criteria} were environmental quality criteria set by European or US environmental authorities for the chemical stressor or nutrient considered (*Appendix 1-Table 1*), and $[Compound_i]_{treatment}$ were the concentrations of the chemical stressor or nutrient at the treatment or impacted sites. When multiple stressors or nutrients were reported, we used the standardized

intensity of the stressor or nutrient corresponding to the highest standardized intensity for the rest
 of the analyses.

We used consistent sources for the environmental guality criteria as much as possible. For 508 chemicals, we relied primarily on quality criteria from the European Chemical Agency (ECHA) 509 and United States Environmental Protection Agency (USEPA) that use standardized procedures 510 across aquatic and terrestrial realms based on ecotoxicological data. For nutrients, we relied 511 mostly on European Water Framework Directive (WFD) benchmarks. Using various sources for 512 those quality criteria was inevitable due to the high number of chemicals and the various way the 513 authors reported stressor or nutrient levels in individual studies. When we could not find quality 514 criteria for the stressors or nutrients considered in the studies in our main sources, we relied 515 on the authors' statements and expert knowledge regarding their stressor or nutrient levels (e.g. 516 recommended application rates of pesticides, citation for ecotoxicological data, or synthesis studies, 517 (Appendix 1-Table 1)). Despite this, the final datasets were all dominated by similar sources for 518 standardizing stressor and nutrient intensity levels: thresholds from ECHA or USEPA for 80 and 519 90% of observations in the stressor-diversity and stressor-abundance datasets, respectively, and 520 for nutrients, thresholds from WFD for 100 and 75% of observations in the nutrient-diversity and 521

⁵²² nutrient-abundance datasets, respectively.

⁵²³ Overall effects of chemical stressors and nutrient enrichment: first-level meta-⁵²⁴ analysis

We first tested the differences between the effects of chemical stressors and nutrient enrichment 525 on decomposer diversity, abundance and litter decomposition responses by quantifying the grand 526 mean effect sizes on the three response variables (first level meta-analysis). Three separate meta-527 analyses were conducted, one for each response variable, and included the type of driver (stressors 528 or nutrients) as a categorical moderator, and a random effect of the case study. We used a weighted 529 meta-analysis giving more weight to effect sizes derived from studies with larger sample sizes. 530 Weights were the inverse of the variance in z-transformed correlation coefficients (Viechtbauer 531 **2010**) Publication bias was evaluated using funnel plots with environmental change driver type as 532 covariate. The intercepts from Egger's regressions (standardized effect size vs. precision = 1/SE) 533 were inspected for significant deviation from zero that would indicate publication bias (Koricheva 534 et al., 2013). Residual plots were used to detect strong deviation from normality and outliers. 535 We estimated the grand mean effect sizes and compared the effect of chemical stressors and of 536 nutrients using Wald-type chi-square tests. The rma.mv() function of the R package metafor was 537 used (R Core Team, 2018: Viechtbauer, 2010). 538

Relationship between biodiversity and decomposition: Structural equation mod elling

An SEM was fitted to estimate the relationship between decomposer diversity or abundance and 541 litter decomposition responses to environmental change drivers while controlling for the joint 542 influence of stressor or nutrient intensity and categorical covariates. We used piecewise SEM 543 (Lefcheck, 2016) estimating two linear mixed effect models, one for decomposition (z_{LP}) and one 544 for decomposer diversity or abundance responses (z_p) , with a random effect of the case study 545 on the intercepts. These two sub-models embedded in the piecewise SEM were the second-level 546 meta-analyses in our hierarchical approach. The random effect structure, weighting approach and 547 variance structure were coded with the R package nlme (Pinheiro et al., 2018) in a way that fully 548 reproduced the meta-analysis approach of weighting and of known residual variance (Viechtbauer, 549 2016): 550

 $z_{LD} \sim z_B + ECD_{intensity} + study type, random = ~1|Case study/ID$

 $z_B \sim ECD_{intensity} + study type + taxonomic group (+diversity metric), random =~ 1|Case study/ID$

This SEM was tested separately for each of four datasets: Stressors – Biodiversity; Stressors - Abundance; Nutrients – Biodiversity and Nutrients – Abundance datasets. The influence of the diversity metric (diversity indices versus taxa richness) was tested in the Biodiversity datasets only. We initially considered more complex model structures, but were unable to use them for analysis due to data limitations (in particular the effect of the ecosystem type and of interactions between our covariates).

Outliers, relationships between covariates, and non-linear patterns between continuous covari-559 ates were explored graphically. Studies often reported different decomposer diversity or abundance 560 values for the same litter decomposition (e.g. when several taxonomic or functional groups were 561 reported in the same litterbag). This variability could have affected the model estimates. We 562 thus used data resampling to account for duplicated effect sizes on litter decomposition in the 563 analyses. A stratified resampling was conducted, where for each duplicated value of effect size on 56/ decomposition, one randomly selected effect size on biodiversity was kept at each out of 1,000 565 iterations. The models were fitted for each data resampling iteration, and we averaged model 566 estimates and statistics across iterations and used the means as final values (path coefficients and 567 standard error of the path and intercepts. Chi-square statistics and AICs). 568

Goodness-of-fit of the SEMs was assessed using directed separation tests based on the Fisher's 569 C statistic. We used mediation tests to explore the significance of the path between decomposer 570 diversity or abundance and litter decomposition based on the Fisher's C statistic of SEM that did not 571 include the biodiversity-mediated path (Lefcheck, 2016: Shipley, 2009). We calculated the P-value 572 associated with the mean Fisher's C statistic across data resampling iterations (P-value < 0.05 573 indicated poor model fit). The AICs of models with and without the biodiversity-mediated paths 574 were further compared using averaged AICs across data resampling iterations. We considered 575 the biodiversity (or abundance) path to be consistent with the data when the SEM without the 576 biodiversity-path had P-value < 0.05 (poor fit) and was not associated with a better AIC value 577 (i.e. lower than 2 units) than the SEM including the biodiversity path. Residuals from the two sub-578 models of each SEM were graphically evaluated for strong departure to normality and relationship 579 with the fitted values (*Duffy et al., 2015*). For these analyses, we averaged the residuals across 580 data resampling iterations for each observation. We finally compared the relative magnitude of 58 the biodiversity-mediated path versus the direct path from stressor or nutrient intensity to litter 582 decomposition based on the mathematical product of the standardized path coefficients (Grace, 583 2006). 584

585 Moderator analyses: Second-level meta-analyses

In order to quantify the influence of the categorical (study type, taxonomic group and diversity metrics) and continuous (environmental change intensity) moderators on the three response variables, we further analyzed the results of the second-level meta-analyses (i.e. the sub-models embedded in the SEMs). The data resampling used in the SEM was no longer necessary, because there were no repeated values of decomposition matching different decomposer diversity or abundance measurements in this univariate approach. We quantified the effects of the different moderators based on the Wald-type chi-square tests derived with the R package metafor (*Viechtbauer, 2010*).

593 Sensitivity analyses

We finally tested the robustness of the results to the approximation of standard deviations, the presence of extreme values, and the metric of effect size used. The analyses were re-run with datasets that did not include the effect sizes for which we approximated standard deviations, for datasets that did not include extreme values of effect sizes (values beyond the whiskers of boxplots i.e. below quantile 1 minus 1.5 times the interquartile range or above quantile 3 plus 1.5 times the interquartile range). Finally, we calculated log-response ratios instead of correlation coefficients as effect sizes and re-run the analyses.

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Authors contribution

- 607 All authors conceived the project; L.B. collected the data, performed the analyses and wrote the
- manuscript; all authors discussed the results and contributed to the manuscript text.

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Appendix 1

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Supplementary Methods

Search strategy

We collected data from published papers reporting the effect of various global change drivers on both decomposition rates and decomposer communities. The search strategy first involved the selection of a relevant search term combination. We compared different search term combinations based on the number of studies retrieved, their potential relevance (based on screening the titles in the search), and on maximizing the retrieval of pre-identified papers that fully matched the inclusion criteria. We used search terms of previous meta-analyses and literature reviews (Garcia Palacios et al., 2016; Covich et al., 2004; Srivastava et al., 2009). The following search terms were used to identify studies looking at the impact of various global change drivers on both decomposition rates and decomposer communities. The search was done on ISI Web Of Science on November 17th 2017 and retrieved 2,536 records.

TS=("global change" OR "environmental change" OR disturbance* OR stress* OR "climat* change" OR drought OR temperature* OR warming OR heat* OR precipitation* OR rain* OR flood* OR irrigation OR moisture OR watering OR fire OR "carbon dioxide" OR CO2 OR acidification OR "nitrogen deposition" OR "nutrient deposition" OR "atmospheric deposition" OR *eutroph* OR fertili* OR "nutrient* enrichment" OR "nutrient pollut*" OR "land-use" OR "landuse" OR "agricultural intensi*" OR desertif* OR pollut* OR pesticide* OR metal* OR "over-exploit*" OR overexploit* OR toxi* OR contamin* OR over-fish* OR invasi* OR alien OR "habitat loss" OR "habitat fragment*" OR "habitat degrad*" OR "habitat destruct*")

AND

TS = ((decomposition OR processing OR breakdown OR decay OR "mass loss") AND (litter OR leaf OR leaves OR bark OR wood))

AND

TS= (("species richness" OR richness OR "number of species" OR "number of taxa" OR "species diversity" OR "taxonomic diversity" OR biodiversity OR Shannon* OR evenness OR "community composition" OR "community structure" OR "functional diversity" OR "trait diversity" OR "functional traits" OR "functional group richness" OR "trait-based") AND (decomposer* OR detritivore* OR *invertebrate* OR microb* OR microorganism* OR bacteri* OR fung* OR archaea OR shredder OR *invertebrate* OR hyphomycete* OR "leafshredding" OR "leaf-eat*" OR "leaf-consum*" "leaf-feed*" OR "litter-feed*" OR "litter-eat*" OR "litter-shredding" OR protozoa* OR protist* OR springtail OR collembol* OR mite* OR acari* OR enchytraeid* OR nematod* OR rotifer* OR isopod* OR earthworm* OR termite* OR microarthropod* OR macroarthropod* OR microfauna OR mesofauna OR macrofauna))

Abstracts were individually screened using the online software Abstrackr (https://abstra ckr.cebm.brown.edu) to identify references matching our inclusion criteria. At the screening step, tags were given to classify studies according to the type of drivers. This step resulted in 384 articles potentially relevant for the meta-analysis, 2,152 abstracts did not match the inclusion criteria (mostly because they were not looking at both decomposition rates and decomposer communities responses to global change, or because they manipulated decomposer communities directly).

We refined the scope of the analysis to focus on two contrasting types of drivers for which we had opposing hypotheses relative to their effects: chemical stressors and nutrient enrichment. These two drivers had a high number of studies, were represented by aquatic and terrestrial studies, and had similar designs (gradients or control vs. treatment with increased concentrations in chemical stressors or nutrients). With the refined scope, 112

studies were potentially relevant, and 272 studies were excluded based on the tags defined at the screening step.

Figure 1 reports the PRISMA diagram describing the different steps to assemble our datasets. After full text screening of the 112 potentially relevant papers, 61 papers verified our inclusion criteria and reported data that we could extract for the meta-analysis. For the SEM analysis, 2 papers were further excluded because some data needed for the models were missing (typically the levels of nutrients or stressors).



Appendix 1 Figure 1. PRISMA plot describing the data collection steps of the meta-analysis. SEM = structural equation modelling.

Environmental quality standards

Appendix 1 Table 1. Environmental quality criteria for stressors and nutrients. Quality criteria were used to standardized the intensity levels of the different chemical stressors across studies included in the meta-analysis.

System	Chemical or Nutrient	Unit1	Unit2	Quality Criteria	citation
aquatic	fungicide: pyrimethanil	µg/l	-	0.69	Abelho, M., Martins, T. F., Shinn, C., Moreira-Santos, M. & Ribeiro, R. Effects of the fungicide pyrimethanil on biofilm and organic mat- ter processing in outdoor lentic mesocosms. Ecotox- icology 25, 121–131 (2016).
aquatic	fungicide: tebuconazole	µg/I	-	0.10	https://echa.europa.eu/d ocuments/10162/41e9d7a a-4559-f904-9cb5-0a0d5f 0d6445
aquatic	As	µg/l	-	13.00	https://echa.europa.eu/b rief-profile/-/briefprofile/ 100.028.316

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aquatic

terrestrial

terrestrial

terrestrial

terrestrial

terrestrial

terrestrial

terrestrial

terrestrial

AI	µg/l	-	87.00	https://www.govinf o.gov/content/pkg/ FR-2018-12-21/pdf/
Cu	µg/l	-	10.10	2018-27/45.pdf https://echa.europa.eu/b rief-profile/-/briefprofile/
Zn	µg/l	-	20.60	https://echa.europa.eu/b rief-profile/-/briefprofile/
Fe	µg/l	-	1000.00	https://www.epa.gov/wqc/ national-recommended-w ater-quality-criteria-aquat ic-life-criteria-table
Mn	µg/l	-	1000.00	https://www.epa.gov/wqc/ national-recommended-w ater-quality-criteria-aquat ic-life-criteria-table
Hg	µg/l	-	0.06	https://echa.europa.eu/b rief-profile/-/briefprofile/ 100 028 278
Cd	µg/l	-	0.19	https://echa.europa.eu/b rief-profile/-/briefprofile/
insecticide: chlorpyrifos	µg/l	-	0.08	https://www.epa.gov/wqc/ national-recommended-w ater-quality-criteria-aquat
phenanthrene	μg/l	-	51.40	Wu, JY. et al. Develop- ment of water quality cri- teria for phenanthrene and comparison of the sensitiv- ity between native and non- native species. Environ- mental Pollution 196, 141-
Zn	mg/kg	-	117.80	146 (2015). https://echa.europa.eu/b rief-profile/-/briefprofile/
Cd	mg/kg	-	1.80	100.028.341 https://echa.europa.eu/b rief-profile/-/briefprofile/
Hg	mg/kg	-	9.30	https://echa.europa.eu/b rief-profile/-/briefprofile/
Pb	mg/kg	-	186.00	https://echa.europa.eu/b rief-profile/-/briefprofile/ 100.028.273
Cu	mg/kg	-	106.35	https://echa.europa.eu/b rief-profile/-/briefprofile/ 100.124.825
Zn	mg/kg	-	35.60	https://echa.europa.eu/b rief-profile/-/briefprofile/ 100.028.341
Ni	mg/kg	-	29.90	https://echa.europa.eu/b rief-profile/-/briefprofile/ 100.028.283
Mn	mg/kg	-	3.40	https://echa.europa.eu/b rief-profile/-/briefprofile/ 100.028.277
Hg	µg/kg	-	22.00	https://echa.europa.eu/b rief-profile/-/briefprofile/ 100.028.278
Pb	mg/kg	-	212.00	https://echa.europa.eu/b rief-profile/-/briefprofile/ 100.028.273
Cd	mg/kg	-	0.90	https://echa.europa.eu/b rief-profile/-/briefprofile/ 100.028.320
insecticide: chlorpyrifos	kg/ha	-	1.25	Iwai, C. B. & Noller, B. Ecotoxicological assess- ment of diffuse pollution using biomonitoring tool for sustainable land use in Thailand. Journal of Environmental Sciences 22, 858–863 (2010).

terrestrial	insecticide: endosulfan	kg/ha	-	1.25
terrestrial	herbicide: atrazine	kg/ha	-	1.88
terrestrial	insecticide: carbofuran	kg/ha	-	31.25
aquatic	pesticide mixture	arbitrary	-	1.00
terrestrial	herbicide: glyphosate	kg/ha	-	4.32
terrestrial	herbicide: simazine	kg/ha	-	0.10
aquatic	pesticide mixture	sum or max of TU (toxic units)	-	-3.50
aquatic	DIN	mg/l	Ν	3.05
aquatic	NH4+	mg/l	NH4	0.10
aquatic	NO3	mg/l	NO3	10.00

Iwai, C. B. & Noller, B. Ecotoxicological assessment of diffuse pollution using biomonitoring tool for sustainable land use in Thailand. Journal of Environmental Sciences 22, 858-863 (2010). Iwai, C. B. & Noller, B. Ecotoxicological assessment of diffuse pollution using biomonitoring tool for sustainable land use in Thailand. Journal of Environmental Sciences 22, 858-863 (2010). Iwai, C. B. & Noller, B. Ecotoxicological assessment of diffuse pollution using biomonitoring tool for sustainable land use in Thailand. Journal of Environmental Sciences 22, 858-863 (2010). Talk, A. et al. Effects of multiple but low pesticide loads on aquatic fungal communities colonizing leaf litter. Journal of Environmental Sciences 46, 116-125 (2016). European Food Safety Authority (EFSA). Conclusion on the peer review of the pesticide risk assessment of the active substance glyphosate. EFSA Journal 13, (2015). https://ec.europa.eu/food/ plant/pesticides/eu-pestici des-database/public/?even t=activesubstance.detail&l anguage=EN&selectedID= 1853 Schäfer, et.al., 2007. Effects of pesticides on community structure and ecosystem functions in agricultural streams of three biogeographical regions in Europe. Science of The Total Environment 382, 272-285. Ministère de l'Environnement, de l'Énergie et de la Mer. Guide technique Relatif à l'évaluation de l'état des eaux de surface continentales (cours d'eau, canaux, plans d'eau). (2016). Ministère de de l'Environnement, l'Énergie et de la Mer. Guide technique Relatif à l'évaluation de l'état des eaux de surface continentales (cours d'eau, canaux, plans d'eau). (2016). Ministère de l'Environnement, de l'Énergie et de la Mer. Guide technique Relatif à l'évaluation de l'état des eaux de surface continentales (cours d'eau, canaux, plans d'eau). (2016).

aquatic	NO2	mg/l	NO2	0.10	Ministère de l'Environnement, de l'Énergie et de la Mer. Guide technique Relatif à l'évaluation de l'état des eaux de surface continen- tales (cours d'eau, canaux, plans d'eau) (2016)
aquatic	Total_N	mg/l	Ν	0.67	US EPA, O. Water Quality Criteria. US EPA (2013). Available at: https://www. epa.gov/wqc. (Accessed: 7th January 2010)
aquatic	SRP	mg/l	PO43	0.10	Guide technique Relatif à l'évaluation de l'état des eaux de surface continen- tales (cours d'eau, canaux, plans d'eau). (Ministère de l'Environnement, de l'Énergie et de la Mer, 2016)
aquatic	Total_P	mg/l	Ρ	0.05	Guide technique Relatif à l'évaluation de l'état des eaux de surface continen- tales (cours d'eau, canaux, plans d'eau). (Ministère de l'Environnement, de l'Énergive et de la Mer, 2016).
terrestrial	N deposition	kg/ha/yr	Ν	20.00	Pardo et al. 2011 (Ecolog- ical Applications); derived critical loads (i.e. level of deposition below which no detrimental ecological ef- fect occurs over the long term according to current knowledge) from empiri- cal data for various (plant)
terrestrial	P fertilization	kg/ha/yr	Ρ	35.00	species and ecosystems Amery, F. & Schoumans, O. F. Agricultural phospho- rus legislation in Europe. (2014).



Appendix 2 Figure 1. Assessment of publication bias. Stressors: Funnel plots of each response variables (decomposer diversity, abundance and decomposition) in the two datasets (stressors - diversity and stressors - abundance). Meta-analytic models included the effect of stressor intensity (standardized levels) as a covariate.



Appendix 2 Figure 2. Assessment of publication bias. Nutrients: Funnel plots of each response variables (decomposer diversity, abundance and decomposition) in the two datasets (stressors - diversity and stressors - abundance). Meta-analytic models included the effect of nutrient intensity (standardized levels) as a covariate.

947 948 949 Appendix 2 Table 1. Assessment of Publication bias. Results from Egger's regressions showing the intercepts, standard error (SE) and P-value of regressions between standard normal deviate of each response variable (effect sizes) and the inverse of their standard errors. Models also included stressor or nutrient intensity as a covariate.

Dataset	Variable	Publication bias P	Publication bias	Intercept	SE
Stressors - Biodiv	Biodiversity	0.10	no	-1.36	0.83
Stressors - Biodiv	Decomposition	0.58	no	-1.07	1.94
Stressors - Abdc	Abundance	0.14	no	-1.49	1.02
Stressors - Abdc	Decomposition	0.68	no	-0.67	1.60
Nutrients - Biodiv	Biodiversity	0.37	no	0.76	0.86
Nutrients - Biodiv	Decomposition	0.19	no	3.35	2.55
Nutrients - Abdc	Abundance	0.08	no	1.21	0.70
Nutrients - Abdc	Decomposition	<.001	pub. bias	5.31	1.45

Meta-analysis - First level: overall mean effects

Appendix 2 Table 2. First level meta-analysis comparing the effects of chemical stressors and nutrient enrichment. Results of Wald-type chi-square tests comparing the grand mean effect sizes of the three response variables (decomposer diversity, abundance and litter decomposition) between chemical stressors and nutrient enrichment.

Response	QM	df	n	P-value
Diversity	25.647174	2	174	<0.001
Abundance	7.916468	2	424	0.019
Litter decomposition	17.611818	2	165	<0.001

SEM analysis

Appendix 2 Table 3. Summary table of structural equation modelling (SEM) analysis.

Unstandardized path coefficients from SEMs for the four datasets: Stressors - Biodiversity (Biodiv), Stressors - Abundance (Abdc), Nutrients - Biodiversity and Nutrients, Abundance. SEMs also incorporated categorical predictors (study type, taxonomic group and diversity metric, see Methods).

Dataset	Response	Predictor	Estimate	SE	Crit.Value	df	P-value
Stressors - Biodiv	Decomposition	Diversity	0.42	0.17	2.50	19	0.022
Stressors - Biodiv	Decomposition	Stressor intensity	-0.02	0.04	-0.47	19	0.643
Stressors - Biodiv	Diversity	Stressor intensity	-0.10	0.04	-2.44	18	0.025
Stressors - Abdc	Decomposition	Abundance	0.24	0.08	2.97	25	0.007
Stressors - Abdc	Decomposition	Stressor intensity	-0.01	0.03	-0.41	25	0.683
Stressors - Abdc	Abundance	Stressor intensity	0.00	0.05	0.03	25	0.977
Nutrients - Biodiv	Decomposition	Diversity	0.01	0.11	0.06	20	0.951
Nutrients - Biodiv	Decomposition	Nutrient intensity	-0.08	0.06	-1.21	20	0.239
Nutrients - Biodiv	Diversity	Nutrient intensity	-0.25	0.07	-3.51	19	0.002
Nutrients - Abdc	Decomposition	Abundance	0.08	0.10	0.76	44	0.451
Nutrients - Abdc	Decomposition	Nutrient intensity	-0.12	0.05	-2.16	44	0.037
Nutrients - Abdc	Abundance	Nutrient intensity	-0.06	0.06	-1.00	44	0.321

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Meta-analysis - Second-level: categorical moderators

Appendix 2 Table 4. Main effects of categorical predictors on decomposer diversity, abundance and decomposition in the four datasets: Stressors - Biodiversity (Biodiv), Stressors - Abundance (Abdc), Nutrients - Biodiversity and Nutrients, Abundance. Results are QM statistics and associated *P*-values of the second-level meta-analyses.

Dataset	Response	Predictor	QM	P-value
Stressors - Biodiv	Diversity	Taxonomic group	4.80	0.028
Stressors - Abdc	Abundance	Taxonomic group	10.10	0.001
Nutrients - Biodiv	Diversity	Taxonomic group	12.77	< 0.001
Nutrients - Abdc	Abundance	Taxonomic group	4.53	0.033
Stressors - Biodiv	Diversity	Study type	1.89	0.169
Stressors - Abdc	Abundance	Study type	0.92	0.338
Nutrients - Biodiv	Diversity	Study type	0.24	0.625
Nutrients - Abdc	Abundance	Study type	0.98	0.323
Stressors - Biodiv	Diversity	Diversity metric	1.67	0.196
Nutrients - Biodiv	Diversity	Diversity metric	2.35	0.125
Stressors - Biodiv	Decomposition	Study type	0.16	0.693
Stressors - Abdc	Decomposition	Study type	1.85	0.174
Nutrients - Biodiv	Decomposition	Study type	2.69	0.101
Nutrients - Abdc	Decomposition	Study type	0.18	0.674

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Sensitivity analyses

Influence of approximating standard deviations

When studies did not report standard deviations associated with the mean decomposer diversity or abundance or the mean decomposition rates, we used linear approximations to estimate the variance based on our data (see Methods). We tested the influence of those approximations on the final results by running the structural equation modelling (SEM) analysis without those effect sizes for which standard deviations were approximated. Overall the same patterns were found showing that approximating missing standard deviations had limited effects on the final SEM results.

Appendix 3 Table 1. Results of mediation tests from structural equation modelling (SEM) analysis based on data without approximated standard deviations. C statistic and associated *P*-value for SEM without the path from biodiversity or abundance to decomposition for the four datasets: Stressors - Diversity, Stressors - Abundance, Nutrients - Diversity and Nutrients - Abundance. Δ-AIC is the difference in AIC score between models with and without biodiversity- or abundance-mediated effects.

Dataset	C statistic	df	P-value	∆-AIC	no.studies	n
Stressors, Biodiv	12.42	6	0.053	-8.32	16	58
Stressors, Abdc	10.15	4	0.038	-6.82	23	216
Nutrient, Biodiv	13.33	6	0.038	-1.46	21	67
Nutrient, Abdc	3.82	4	0.432	-0.12	32	127

Appendix 3 Table 2. Summary table of structural equation modelling (SEM) analysis based on data without approximated standard deviations. Standardized (Std.Est.) and unstandardized

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(Estimate) path coefficients from SEMs for the four datasets.

Dataset	Response	Predictor	Std.Est.	Estimate	SE	Crit.Value	df	<i>P</i> -value
Stressors - Biodiv	Decomposition	Diversity	0.52	0.50	0.16	3.16	12	0.008
Stressors - Biodiv	Decomposition	Stressor intensity	-0.26	-0.05	0.03	-1.54	12	0.148
Stressors - Biodiv	Diversity	Stressor intensity	-0.39	-0.08	0.04	-1.89	11	0.085
Stressors - Abdc	Decomposition	Abundance	0.40	0.27	0.09	2.91	19	0.009
Stressors - Abdc	Decomposition	Stressor intensity	-0.11	-0.02	0.03	-0.77	19	0.450
Stressors - Abdc	Abundance	Stressor intensity	0.08	0.03	0.06	0.46	19	0.649
Nutrients - Biodiv	Decomposition	Diversity	-0.04	-0.04	0.12	-0.35	10	0.732
Nutrients - Biodiv	Decomposition	Nutrient intensity	-0.31	-0.14	0.09	-1.52	10	0.161
Nutrients - Biodiv	Diversity	Nutrient intensity	-0.49	-0.23	0.10	-2.39	9	0.040
Nutrients - Abdc	Decomposition	Abundance	0.05	0.04	0.13	0.33	29	0.742
Nutrients - Abdc	Decomposition	Nutrient intensity	-0.26	-0.12	0.06	-1.91	29	0.066
Nutrients - Abdc	Abundance	Nutrient intensity	-0.20	-0.10	0.07	-1.40	29	0.173

Influence of extreme values

We re-run our SEMs with datasets excluding extreme values of effect sizes. Extreme values were defined as values exceeding the whiskers of boxplots. Overall we found similar patterns showing that extreme effect sizes had limited effects on the final SEM results.

Appendix 3 Table 3. Results of mediation tests from structural equation modelling (SEM)analysis based on data excluding extreme values of effect sizes. C statistic and associated *P*-valuefor SEM without the path from biodiversity or abundance to decomposition for the four datasets:Stressors - Diversity, Stressors - Abundance, Nutrients - Diversity and Nutrients - Abundance. Δ-AIC isthe difference in AIC score between models with and without biodiversity- or abundance-mediatedeffects.

Dataset	C statistic	df	P-value	∆-AIC	no.studies	n
Stressors, Biodiv	10.18	6	0.117	-6.71	22	94
Stressors, Abdc	7.39	4	0.117	-4.23	27	254
Nutrient, Biodiv	14.80	6	0.022	-4.85	26	93
Nutrient, Abdc	2.74	4	0.603	0.15	35	159

Appendix 3 Table 4. Summary table of structural equation modelling (SEM) analysis based on

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data excluding extreme values of effect sizes. Standardized (Std.Est.) and unstandardized (Estimate) path coefficients from SEMs for the four datasets.

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Dataset	Response	Predictor	Std.Est.	Estimate	SE	Crit.Value	df	P-value
Stressors - Biodiv	Decomposition	Diversity	0.41	0.40	0.18	2.20	18	0.041
Stressors - Biodiv	Decomposition	Stressor intensity	-0.04	-0.01	0.04	-0.24	18	0.814
Stressors - Biodiv	Diversity	Stressor intensity	-0.44	-0.10	0.04	-2.75	17	0.014
Stressors - Abdc	Decomposition	Abundance	0.30	0.24	0.11	2.24	23	0.035
Stressors - Abdc	Decomposition	Stressor intensity	0.05	0.01	0.03	0.35	23	0.731
Stressors - Abdc	Abundance	Stressor intensity	0.00	0.00	0.04	-0.02	23	0.980
Nutrients - Biodiv	Decomposition	Diversity	0.00	0.00	0.11	0.02	19	0.986
Nutrients - Biodiv	Decomposition	Nutrient intensity	-0.18	-0.08	0.06	-1.30	19	0.210
Nutrients - Biodiv	Diversity	Nutrient intensity	-0.53	-0.24	0.07	-3.36	18	0.003
Nutrients - Abdc	Decomposition	Abundance	0.00	0.00	0.09	0.04	37	0.968
Nutrients - Abdc	Decomposition	Nutrient intensity	-0.38	-0.13	0.04	-3.26	37	0.002
Nutrients - Abdc	Abundance	Nutrient intensity	-0.24	-0.09	0.05	-1.73	37	0.092

Influence of the effect size metric

We tested the influence of the metric of effect size selected on the results of the SEMs. Log-response ratios were calculated instead of correlation coefficients and the models were re-run based on those data. The results were partially different from the original analysis. For nutrients, similar patterns were found, however for stressors there was limited support for the biodiversity- and abundance-mediated effects on decomposition responses. We noted extreme values of log-response ratios that may have explained such patterns. Besides, the log-response ratio has a different interpretation compared to correlation coefficients. Log-response ratios are sensitive to the different metrics of diversity and abundance, taxa groups, litter types etc. used across studies included in this meta-analysis. Therefore this result reinforced our choice of correlation coefficients as relevant effect sizes in the present meta-analysis.

Appendix 3 Table 5. Results of mediation tests from structural equation modelling (SEM) analysis based on data using log-response ratio as an effect size. C statistic and associated *P*-value for SEM without the path from biodiversity or abundance to decomposition for the four datasets: Stressors - Diversity, Stressors - Abundance, Nutrients - Diversity and Nutrients - Abundance. Δ -AIC is the difference in AIC score between models with and without biodiversity- or abundance-mediated effects.

Dataset	C statistic	df	P-value	Δ-AIC	no.studies	n
Stressors, Biodiv	4.11	6	0.662	-0.02	22	70
Stressors, Abdc	5.59	4	0.232	-2.22	37	150
Nutrient, Biodiv	8.03	6	0.236	-2.08	14	78
Nutrient, Abdc	3.41	4	0.492	-0.44	21	307

Appendix 3 Table 6. Summary table of structural equation modelling (SEM) analysis based on data using log-response ratio as an effect size. Standardized (Std.Est.) and unstandardized (Estimate) path coefficients from SEMs for the four datasets.

Dataset	Response	Predictor	Std.Est	Estimate	SE	Crit.Value	df	P.value
Stressors - Biodiv	Decomposition	Diversity	0.18	0.12	0.15	0.80	15	0.437
Stressors - Biodiv	Decomposition	Stressor intensity	-0.24	-0.05	0.04	-1.47	15	0.163
Stressors - Biodiv	Diversity	Stressor intensity	-0.35	-0.12	0.03	-4.17	15	0.001
Stressors - Abdc	Decomposition	Abundance	0.14	0.04	0.05	0.86	28	0.396
Stressors - Abdc	Decomposition	Stressor intensity	0.09	0.02	0.04	0.55	28	0.586
Stressors - Abdc	Abundance	Stressor intensity	-0.14	-0.11	0.11	-1.03	28	0.312
Nutrients - Biodiv	Decomposition	Diversity	0.29	0.19	0.10	1.80	14	0.094
Nutrients - Biodiv	Decomposition	Nutrient intensity	-0.15	-0.07	0.08	-0.96	14	0.352
Nutrients - Biodiv	Diversity	Nutrient intensity	-0.20	-0.16	0.07	-2.11	14	0.054
Nutrients - Abdc	Decomposition	Abundance	0.06	0.04	0.06	0.59	42	0.559
Nutrients - Abdc	Decomposition	Nutrient intensity	-0.36	-0.16	0.05	-3.08	42	0.004
Nutrients - Abdc	Abundance	Nutrient intensity	-0.01	0.00	0.08	-0.08	42	0.935

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