

1 **1. Point-by-point reply to the reviewer comments**

2

3 **2. Marked-up manuscript version (text only)**

4

5

6

7

8

9

10

11

12

13

14

15

16

17

18

19

20

21

22

23

24

25

26

27

28

29

30

31

32

33

34

35

36

37

38

39 **1. Point-by-point reply to the reviewer comments:**

40 We thank all three reviewers for their constructive comments and suggestions. In the pages
41 below we respond to each of these separately.

42
43 In the following, reviewer comments are shown in in black, **author responses in red** font. Line
44 numbers given below refer to the revised version of the ms (separate file, not the marked-up
45 version below).

46
47

48 **REVIEWER 1, anonymous**

49

50 **General comments**

51 This article deals with an important aspect of carbon's fate in coastal wetlands in relation to
52 global changes and their impacts on these ecosystems. Indeed, wetlands are receiving a growing
53 attention in the climate change debate in relation to their high capacity to sequester blue
54 carbon. Ecosystems considered in this "global" scale study are mainly tidal marches but some
55 mangroves sites were counted in the selected sites. Authors are assessing OM degradation and
56 transformation, as a proxy of Carbon sequestration using the TBI approach. Thus, authors claim
57 that they provided indirect evidences that rising Temperature and Sea Level and eutrophication
58 will impact the capacity of tidal wetlands to sequester carbon. This work is worthwhile to
59 publish although as authors cautioned, there are limits with the used method (obvious quality
60 differences of Tea-bag OM with "real" plants) and also that they may have missed some influent
61 factors that control OM degradation and sequestration. Introduction was well thought and the
62 methodology was clear however, some choices were not judicious in the context of this study
63 and may need to be reevaluated (see specific comments). The adding of TIDE experimental site
64 was a very interesting. The discussion is well organised but it needs to be shortened.

65

66 **We thank the reviewer for his constructive feedback on our study. As requested, we**
67 **streamlined the discussion where appropriate; however, several required additions have also**
68 **been made, so that the overall length of the Discussion did not substantially change.**

69

70 **Specific comments**

71 I am not a specialist of meta-analysis, therefore I will not comment on the validity or not of the
72 numerical methods, but one thing is sure, analyses need always to rely on field knowledge even
73 if results are "counterintuitive". The discussion is based on two characteristics (k, S) that are
74 related to the quality and the fate of the litter-bags contents (here Tea-bags) which are strongly
75 related to sedimentation dynamic and water velocity. In absence of a clear indication on how
76 sediments (and OM) are behaving in each site, I am concerned about the amalgam in the same
77 meta-analysis different systems in term of hydrological functioning: Salt Marches vs.
78 Mangroves, High tide vs. low tide (in salt marches). For instance, estuarine mangroves receive
79 loads of sediments from rivers whereas Europeans salt marches in open Bays get sediments
80 mainly from the oceans. One way to tackle this concern is to process the same calculations/test
81 s/figures without adding the mangrove sites to the pool of data. Same thing can be done by
82 considering the main origin of sediments (not to confound with OM), without impacted TIDE
83 sites, river presence or not, water velocity, human activities: These factors, of ecological

84 importance, might be those missing to explain some global, or local, differences. If these data
85 cannot be compiled they should at least be discussed.

86
87 We agree with the reviewer that factors other than those assessed in this study might have
88 been influential and could have masked expected results (i.e missing temp effect on k). In
89 accordance with the reviewer's suggestion, we elaborated on this in several sections of the
90 discussion: e.g. 392-395; 399-407.

91
92 Response concerning sedimentary factors:

93 We agree with the reviewer that the different systems we compiled in a single meta-analysis are
94 characterized by potentially important differences in both sediment load and origin. We did not
95 explicitly assess sediment loads of our study sites. However, by distinguishing between
96 minerogenic and organogenic systems (i.e. sediment rich vs. sediment poor systems) in our
97 analyses, we are confident to have already captured the relative importance of sediment load
98 on our response variables. Please note that this categorical factor did not show up to be
99 important in our classification- & regression-tree data mining (CART). Furthermore, our two
100 most important findings (i.e. S decreases with temperature; S is lower in low vs. high elevated
101 zones) are consistent within both minerogenic and organogenic systems.

102
103 We indirectly also addressed sediment origin (riverine vs. marine) by including both estuarine
104 and coastal systems in our study. Specifically, we tested for effects of salinity class (fresh,
105 brackish, salt) on our response variables, with fresh systems far up in the estuary experiencing
106 the lowest marine influence and salt-water systems experiencing the highest marine influence.
107 If sediment origin (riverine vs. marine) had an important influence on our response variables,
108 this should have been reflected in our meta-analyses (i.e. Table 2). That being said, salinity of
109 floodwater and sediment origin can of course not easily be separated in an observational study.
110 Concerning the reviewer's remark on the sediment origin of our mangrove sites, it needs to be
111 noted that those were not estuarine (as assumed by the reviewer) but coastal systems in the
112 present study.

113

114 **REVIEWER 2, Dr J. Keuskamp**

115 **General comments**

116 This paper discusses the control that the soil matrix exerts on the decomposition of organic
117 matter in tidal wetlands. Their large carbon stocks and sensitivity to global change make this a
118 highly relevant topic for scientists and policy makers alike. The paper is well-written and easy to
119 read, while presenting novel data with important conclusions on the relation between
120 decomposition and global change. The usage of a standardised method over a wide range of
121 tidal systems allows for a generalisation to the global scale, making this paper relevant to the
122 broad readership of Biogeosciences. The explorative nature of the experiment also introduced
123 some unavoidable methodological weaknesses. Many of the environmental parameters which
124 are discussed in relation to decomposition are often strongly correlated with tidal regime (i.e.
125 soil temperature, salinity, nutrient status, microbial biomass, and redox status), or latitude (i.e.
126 nutrient limitation, vegetation type). In its current version, the manuscript does not always

127 acknowledge the potentially spurious relation between these factors. While this does not
128 invalidate the main conclusions I would recommend to consider non-causality more carefully
129 when attributing effects to specific environmental parameters.

130
131 We thank Dr Keuskamp for his constructive comments on our study. We agree that correlations
132 between the assessed environmental parameters should be carefully considered in the
133 interpretation of our results. Accordingly, we put more emphasize on this throughout the ms;
134 some examples from different sections:

135
136 **Methods/statistics:**
137 “As we did not expect temperature to be independent of other parameters in this observational study, we constructed
138 a Spearman correlation matrix including the parameters temperature, latitude, tidal amplitude, salinity class, *k*, and *S*.
139 Additionally, we tested for differences in these parameters between marshes and mangroves and sites with mineral
140 and organic soils, using Mann-Whitney U tests (Table 2).”

141
142 **Results:**
143 “Temperature was highly correlated with latitude and tidal amplitude, and temperature was not independent of soil
144 type (mineral/organic) and ecosystem type (marsh/mangrove) (Table 2). The effect of latitude was similarly
145 pronounced as the temperature effect on *S* – and consequently –effects of these two parameters on *S* cannot be
146 separated (Table 2). By contrast, tidal amplitude and soil type did not significantly affect *S*, and the difference in *S*
147 between mangroves and marshes was only marginally significant (Table 2). These findings suggest that the presented
148 temperature effect on *S* occurs to be mainly independent of tidal amplitude and soil type.”

149
150 **Discussion:**
151 “Future experimental work is therefore required in order to further assess the effects of temperature on OM
152 stabilization and to separate temperature from latitudinal and other interacting effects (e.g. as outlined above for *k*)
153 that are difficult to control for in observational studies.”

154
155 The reviewer is specifically concerned about interactions with the parameters tidal regime and
156 latitude.

157 →In terms of describing the tidal regime, we assessed tidal amplitude and, by comparing high
158 and low elevated zones within sites, a relative measure for flooding frequency (i.e. low zones
159 more frequently flooded than high zones). Tidal amplitude did not affect *k* and *S* (Table 3). It
160 showed up as a potentially important predictor in our CART, probably because of its strong
161 correlation with other parameters. However, this result needs to be considered cautiously
162 because splits based on tidal amplitude suggest mixed effects (Fig. S1a).

163 →In terms of flooding frequency (high vs. low elevated zones), we discuss a number of
164 potentially relevant interactions that were mentioned by the reviewer: redox -> 423-434;
165 salinity -> 444; nutrient status -> 452-465. The reviewer makes an important point by
166 mentioning soil temp interactions with tidal regime. We will address this point further below
167 where soil vs air temp differences are discussed.

168 →Changes in nutrient status/limitation and vegetation type with latitude are relevant for the
169 interpretation of the temperature effects on *S* and *k*. We accordingly extended discussion of
170 temperature effects on *k* and *S*: 392-395; 399-407

171
172 The current description of the data-analysis does not describe how the authors have
173 ascertained themselves that underlying assumptions of the statistical tests used were not

174 violated. Where applicable, tests of heterogeneity, normality, and independence should be
175 included, or other tests considered.

176 The reviewer is correct. We revised the description of the statistics and also corrected some
177 statistical analyses:

178

179 Specified assumption checks:

180 “To test for effects of relative elevation (as proxy for relative sea level) on k and S, two-tailed paired t-tests were
181 conducted. Mean values of high and low elevated zones of the 21 sites where tea bags were deployed in both high
182 and low elevation zones were used (n = 21). The absence of outliers and normal distribution of the difference in the
183 independent variable (as assessed visually) assured robustness of paired t-tests. To assess the consistency of potential
184 effects of relative elevation on k and S, one-way ANOVAs were used in each site separately (replication was
185 sufficient in 20 sites). Normal distribution of residuals was assessed visually, Levene’s test was used to test for
186 homogeneity of variance, and data were log-transformed if assumptions were not met. Mann-Whitney U tests were
187 conducted as a non-parametric alternative when log-transformed data did not meet ANOVA assumptions (Table S2).
188 We tested for effects of nutrient enrichment on k and S in the data from the TIDE project site (Massachusetts, US)
189 using two-way ANOVA with enrichment treatment and marsh zone as predictors. When Levene’s test indicated
190 heterogeneous variance (true for k), data were log-transformed, which stabilized variance. Normal distribution of
191 residuals was assessed visually.”

192

193 Corrected statistics/data structure:

194 Statistics in Table 2 (Spearman correlations and U tests) were based on mean values of each
195 site-by-zone combination (n = 51). Sites with observations in two zones were thus
196 overrepresented. In the revised version these statistics are based on site means (n = 30;
197 compare 2.2).

198

199 A related point is that the three sites at the Ebro delta (and the three Maine sites) were
200 considered as different zones of the same site, characterized by different salinities
201 (fresh/brackish/salt marsh). However, we noticed that they are actually as far apart as the two
202 sites in Massachusetts or the three mangrove sites in Panama. For reasons of consistency, they
203 are now considered separate sites. This, however, does not change any of the
204 findings/conclusions previously drawn. We clarify in the Methods that many of our sites are co-
205 located in larger estuarine/coastal regions (line 148 and revised Table 1).

206

207 For example a linear fitting is performed between k and S with temperature, without
208 mentioning testing for residual patterns to uncover non-linearity. As the authors note the
209 relation between decomposition and single parameters are often not linear (L221), in which
210 case the result of a linear model is unreliable.

211

212 The reviewer is correct. A linear effect of temp is not expected. The intention for showing the
213 linear fit was only to better illustrate the significant temp effect on S (as tested/identified with
214 non-parametric Spearman correlation). However, we should not have used linear regression as
215 an additional hypothesis test. → We only use Spearman correlation for this in the revised
216 version. Yet, to better illustrate the temp effects, we still present scatterplots and use curve
217 fitting to illustrate significant temp effects. Indeed, the model with both highest R2 and lowest
218 standard error of estimate describing the significant temp effect on S is not linear but
219 logarithmic (Figure 2)

220

221 Lastly, I would like to add that the strength of the TBI lies in its standardisation. I would
222 therefore recommend to mention the S/k calculated with the standard approach alongside with
223 the re-scaled values calculated with the more aggressive extraction method. This would allow
224 for easy comparison with other data such as the TBI-values from mangroves mentioned in the
225 methods paper. See also below.

226
227 We agree with the reviewer. The same point has been raised by Dr Sarneel in an interactive
228 comment (below). We have prepared a table with all site x zone values for k and S , giving both
229 the original TBI-values and the modified (Table S3; referenced in the ms: L513).

230

231 **Specific comments**

232 L79 and L83-L84 seem largely redundant to me

233

234 Deleted old lines 79-80

235

236 L85-L86 'OM decomposition' is somewhat ambitious as it is not clear whether this refers to
237 decomposition rate (k) or extend (S), please revise.

238

239 The sentence was changed to "Consequently, global changes that might decrease OM preservation in
240 tidal wetland soils not only affect carbon sequestration, but also decrease ecosystem stability against
241 SLR." Obviously, preservation is also affected by decomposition rate and stabilization; however,
242 we do not intend to specify the processes at this stage of the Introduction, but do this further
243 down in the text (i.e. 115-119).

244

245 L117 Although this should have been more explicit in the TBI method paper (Keuskamp et al,
246 2013), the k estimated by TBI is not exactly equivalent to the classical litter bag experiment as it
247 describes the decomposition rate of the hydrolysable fraction and is not calculated over the
248 entire mass. We have therefore adapted k_1 to indicate that this is the k of the most labile
249 fraction, as opposed to k_2 which refers to the decomposition rate of the recalcitrant fraction. To
250 avoid confusion this should be made explicit here.

251

252 We avoid reference to classical litter bag experiments here and instead make the meaning of k
253 clearer in the respective section of the Methods.

254

255 L120 The recalcitrant fraction is also decomposable, albeit a lot slower

256

257 This was poor wording of course → changed to "rapidly decomposable".

258

259 L127 'thereby improving our process-level understanding on how global warming affects
260 carbon turnover' Not sure what this means exactly

261

262 Deleted "process-level".

263

264 L137 I am somewhat surprised that the oxidation of organic matter would be limited by the
265 supply of SO₄ in brackish tidal wetlands. Wouldn't the constant flushing with water replenish
266 SO₄ to saturating levels in brackish/salt water systems?

267
268 Well, it probably depends on how much seawater input the brackish system experiences.
269 Anyhow, our dataset does not actually allow to accurately describe salinity effects on k and S
270 (too imbalanced, low number of fresh systems). We simplified accordingly and only test for
271 effects of temperature, relative sea level, and eutrophication. Compare 120-135, 240-265

272
273 L154 '(i.e. dwarf vs. fringe phenotypes)' Aren't these also Rhizophora vs Avicennia? In that case
274 phenotypes would not be the appropriate description. These mangroves belong to different
275 genera, each with their own properties (soil oxygenation, phenolic compound production, N-
276 content) that are known to influence decomposition.

277
278 In most cases you would assume so, but here both fringe and dwarf are indeed Rhizophora with
279 very few individuals of Avicennia: Please compare: Mckee et al. (2007) Global Ecology and
280 Biogeography, 16, 545–556; Lovelock et al. (2005) Caribbean Journal of Science, Vol. 41, No. 3,
281 456-464, 2005

282
283 L154 'Relative elevation' as relative to what? mean lower tide, mean mean tide? Please specify
284

285 We specified as follows:

286 "In 21 sites, we compared high and low elevated zones, which were characterized by distinct plant species
287 compositions (i.e. different communities in high vs. mid vs. low marshes) or by different stature of
288 mangroves (i.e. dwarf vs. fringe phenotypes). We used relative elevation (i.e. high vs. low elevated zone)
289 as a site-specific proxy for relative sea level. By doing so, we did not capture the actual variability in the
290 tidal inundation regime across our study sites as these vary in absolute elevation and in elevation relative
291 to mean high water."

292
293 L169-170 Decomposition rates depend on soil temperature rather than on air temperature.
294 Others have shown (e.g Piccolo et al. 1993, Reckless et al. 2011) that in tidal wetlands, the soil
295 temperature is strongly determined by inundation regime in which case the accuweather
296 temperature are not an accurate reflection of the decomposition environment. Moreover,
297 inundation regime and temperature effects would be confounded. Could it be shown
298 accuweather estimated temperatures vs measured temperatures so that the reader can see for
299 themselves whether the accuweather approximation suffices?

300
301 Dr Keuskamp brings a valid point here that indeed needs more consideration. Air temperature
302 would obviously diverge from soil temperature depending on factors such as canopy shading or
303 tidal regime and water temperature. As a consequence, air temperature can only approximate
304 the temperature conditions of the actual decomposition environment. However, considering
305 that we stretch a temp gradient of approx. 20°C, we are confident that this would also translate
306 into a profound soil-temperature gradient across our study sites.

307
308 The two studies mentioned by the reviewer, Piccolo et al. 1993 and Ricklefs et al. 2012, present
309 data for un-vegetated tidal flat systems. For marsh systems, we would rather refer to Kirwan et

310 al. (2014). The authors show, that in marshes along the well-studied latitudinal gradient of the
311 US East coast (and we do share a number of sites), soil temp and air temp are highly correlated,
312 while the relationship between soil temp and water temp is far weaker (Kirwan et al. 2014,
313 *Temperature sensitivity of organic-matter decay in tidal marshes*; biogeosciences: **Fig. 2a**).

314
315 In our study sites, we did not continuously measure soil temp over the 3 months of deployment,
316 and thus it is difficult to assess how well soil and air temp were correlated in this study.
317 However, in several of our sites, soil temp was assessed at the time point of insertion and
318 retrieval of bags. We plotted these data against the mean air temp of the day as acquired from
319 the *accuweather* service in **Figure S3**. → We see that generally air temp is a good proxy for soil
320 temp across sites. Yet, there is considerable variability in soil temp not explained by air temp,
321 which would result from the fact that soil temp was assessed in one time point as opposed to
322 mean air temp of a single day and of course from other factors, such as distance of weather
323 station from site, shading, influence of water temp etc..

324
325 →We agree with the reviewer that this needs to be stated and discussed in the manuscript.
326 Accordingly, we put more emphasize on this throughout the ms; some examples from different
327 sections:

328
329 **Methods:**
330 “It needs to be noted here, that top-soil temperature would differ from air temperature depending on factors such as
331 canopy shading or tidal regime and water temperature. As a consequence, air temperature can only approximate the
332 temperature conditions of the actual decomposition environment (Fig. S3).”

333
334 **Discussion:**
335 “The present study used air temperature as a proxy for top-soil temperature. Thus, the temperature regime of the
336 decomposition environment was only approximated, which certainly would have weakened a significant relationship
337 between temperature and k. However, following typical Q10 values for biological systems of 2-3 (Davidson and
338 Janssens, 2006), k should have at least doubled over the gradient of $\Delta T > 15^\circ\text{C}$; yet our data do not even show a
339 tendency of an effect ($r_s = 0.02$; Table 2). We therefore propose that other parameters exerted overriding influence
340 on k, mainly masking temperature effects, and have not been captured by our experimental design. This notion is in
341 line with the fact that studies conducted at ...”

342
343 Lastly, we want to stress a related point here: “low” and “high” in the figure legend of Fig.S3
344 refer to the low and high elevated zones within the systems. A paired t-test comparing the
345 difference of air temp and soil temp between the paired high and low elevated zones indicates
346 no significant effect of zone ($p = 0.563$). This shows that differences between air and soil temp
347 were not consistently more pronounced in either the low or the high elevated zones.
348 Additionally, soil temp was not significantly affected by zone ($p = 0.342$). One of our main
349 findings, that S is consistently lower in low vs. high zones, is consequently not temperature
350 affected (i.e. S was significantly reduced in 14 of 20 sites, and the opposite was observed in
351 none of the sites (Table S2).

352
353 L176 Pepsico, to my knowledge the bags are produced by Lipton, which is a Unilever brand.

354
355 Unilever belongs to PepsiCo, but of course the tea is produced by Unilever. PepsiCO was deleted
356 in order to avoid confusion.

357
358 L180 Were the reference bags dried at 70oC prior to mass determination?
359
360 This may be a misunderstanding: reference bags were used to determine a mean value of the
361 empty nylon bag itself without contents. I do not know if that one has always been dried,
362 however, empty-bag weights were very consistent among labs. →By contrast, initial tea-content
363 weights showed some variability across the involved labs. I also noticed that some labs, after
364 drying at 70°C, used desiccators, in which the material could cool down without sucking
365 moisture, before weighing and some didn't. I therefore assessed if potential moisture
366 differences of the initial tea material or differences in the amount of the initial material could
367 have affected S or k. However, there was no relationship between green initial weight and S
368 ($r^2=0.0003$; $p=0.936$) and no between rooibos initial and k ($r^2=0.005$; $p=0.728$).

369
370 L198-L200 It could well be that the method described is a more accurate operationalisation of
371 the labile (non-hydrolysable) fraction. Redefining the labile fraction and the consequential shift
372 in S, and rescaling of k, may however lead to misunderstandings when the results of this study
373 are used in comparisons with other TBI experiments. I would therefore suggest to provide the
374 TBI S/k values calculated according to protocol alongside the obtained S/k values obtained by
375 the revised protocol.

376
377 We agree. We added a table with the original TBI values accordingly (see comment further up).

378
379 L220-L250 Would you be able to indicate whether potential violations of the assumptions
380 underlying the statistical tests were assessed? For example, were the residuals of the ANOVA
381 procedure tested for normality / homogeneity of variance?

382
383 This was indeed missing. We added these missing details to the Methods, see comment further
384 up.

385
386 L250 It is critical to this conclusion that air temperature is a good proxy of soil temperature
387 (see earlier remark). The interaction between temperature effect and tidal position reinforces
388 the suspicion that this is not the case.

389
390 We agree with the first half of this remark (see addressed further up), but not with the second.
391 That is, there is no clear interaction between tidal position and temperature: Temperature
392 seems to affect k in mesotidal systems (tidal amp >2.1m) with k higher in sites with temp
393 >14.5°C; however, this apparent temp effect is inconsistent within this group of mesotidal
394 systems. That is, sites with temp >18.2°C show lower k than those sites with temp <18.1°C.
395 Please note comment further up: temp did not differ between high and low elevated zone,
396 neither did the temp difference between soil and air.

397
398 L314 As also noted in L313, the absence of a temperature effect is very unusual. Could the
399 authors rule out the possibility that this is due to a mismatch between soil and air temperature?

400

401 We stretch large gradients of approx. 20°C for both soil and air temp, and there is not even the
402 slightest tendency of a temp effect on k (Spearman's rank coefficient = 0.02; Figure 2), while S is
403 strongly affected. It therefore seems that that other factors exert overriding control over k and
404 more strongly mask temp effects on k than on S. Yet, we agree on the need to discuss the
405 methodological inaccuracy in determining temp of the decomposition environment, and we
406 addressed this point (see comment further up).

407
408 We want to stress a related point here concerning the missing temp effect on k: In order to
409 address remarks by Reviewer 3 and demonstrate the usefulness of the TBI method for tidal
410 wetlands, we took a separate look at the data of the North American East coast latitudinal
411 gradient along which previous studies have shown clear temp effects on decomposition
412 processes and microbial biomass (Blum et al. 2004; Kirwan et al. 2014; Mozdzer et al. 2014).
413 Species composition of these marshes is quite constrained (i.e. *Spartina alterniflora* dominated)
414 reducing confounding effects induced by differences in vegetation. Along this gradient, we
415 clearly see an increase in S and also the expected decrease in k, although temp explains more
416 variability for S (Fig. S2). We added this figure to the manuscript in order to illustrate that temp
417 effects on k can be identified on the regional scale, but not on the global scale with more
418 confounding factors.

419
420 The effect of temp on k at the regional scale but the missing effect at global scale is also in
421 agreement with the just recently published article on *Early stage litter decomposition across*
422 *biomes* by Ika Djukic and others. (Although they did not assess specifically k and S in their study
423 using the TBI tea materials, they simply assessed mass loss of the two materials). Across biomes,
424 climate (temp and precipitation) had no effect on break down; however, within biomes the
425 effect was strong.

426
427 L332 I would recommend discussing potential confounding of temperature effects with other
428 changes in decomposition matrix (e.g. nutrient availability, redox status, vegetation, salinity).
429 With respect to k, such reservations are made in L323/L329, but are absent here.

430
431 We agree with the reviewer and added similar considerations for the discussion on temp effects
432 on S (400-407).

433
434 L351 Can this be generalised to continuously submerged parts of the soil? The TBI is at a
435 relatively low depth, where tidal pumping may cause increased influx of oxygen during tidal
436 subsidence. Especially in tannin-rich mangrove systems, temporal oxygenation may make a
437 large difference by allowing breakdown of phenolic compounds (see also Freeman et al, 2001)

438
439 We agree with the reviewer. We elaborated our discussion on expected redox effects, also with
440 respect to comments by Reviewer 3 (427-435, 515-521).

441
442 L445 In mangrove TBI experiments that I have conducted S values have always been positive,
443 and I am somewhat puzzled by the large difference. Negative S values could also be caused by
444 loss of recalcitrant particles as I have observed when using teabags in open water. Did you have
445 any indications that this has taken place here?

446
447 We were puzzled as well when realizing that so many values were lower than they should be.
448 Indeed, the FL mangrove values you report in Keuskamp et al 2013 are considerably higher.
449 That's also when I decided to check whether the quality of the material had changed.
450 →No, I am not aware of loss of particles from the bags in situ. In fact, in a recent study
451 (microcosm study, Wadden Sea) we used the new tea bags (those without nylon mesh) that
452 wouldn't allow for loss of material through the mesh. Also with these bags, we had ~11%
453 negative values (Hao Tang, Peter Mueller et al. unpublished data), comparable to what we
454 found in some Wadden Sea marshes in the present study using nylon mesh bags.
455 →Comparing our results to those reported in Djukic et al. (2018), it becomes clear that negative
456 S values occur less frequently across terrestrial systems, however, are not negligible either.

457
458 Technical corrections L74 Earth? Not sure if this should be with a capital E

459 L77 Separate SRL from citations

460 L94-98 This sentence is very hard to read. Split.

461 L346 add 'in' before 'tidal wetlands'

462 Thanks, technical corrections have been made.

463

464 **REVIEWER 3, anonymous**

465 Mueller et al. conducted decomposition experiments using tea bags based on a standardized
466 approach developed by Keuskamp et al. (2013), across different marsh and mangrove sites in
467 order to cover a gradient in temperature, inundation regime, etc. While such cross-ecosystem
468 studies have a high potential, I feel the impact of this dataset in terms of new insights is
469 relatively limited. The dataset can be published but I feel the impact of the conclusions should
470 be toned down somewhat – the manuscript does not really deliver what the title suggests. The
471 dataset should be publishable, but it needs a more critical discussion and should provide the
472 readers with a more complete overview of the caveats and assumptions used in the TBI
473 approach, so that the readers can better assess what can and cannot be deduced from these
474 data. My main point is that the TBI index – both the original and the modified protocol
475 suggested here – has plenty of limitations and it remains an operationally defined procedure,
476 with several assumptions that are open to discussion. In addition, we are not looking at
477 mineralization of in situ produced material hence some interactive effects will be missed in this
478 approach; results should not be over-interpreted or generalized.

479 We thank the reviewer for his critical and constructive feedback on our work. We have revised
480 our ms, particularly the discussion part (4.4 Methodological considerations; 4.5 Implications), in
481 order to provide the reader with a more complete overview of the assumptions and limitations
482 involved with the TBI approach. We have provided more detailed responses below regarding
483 the specific comments raised by the reviewer.

484

485 **Specific suggestions**

486 L55: “stabilization was 29% lower”: this does not mean much if you do not define stabilization
487 here, it can be interpreted in different ways. For me this remains a somewhat problematic proxy
488 (see further comments).

489
490 We agree with the reviewer’s concern and specified the parameter in the abstract (53-55). The
491 second part of the comment will be addressed further below.

492
493 L60-61: data from the eutrophication experiment: would not extrapolate this to ‘high sensitivity
494 to global change’. Eutrophication will also affect the nutrient content of locally produced
495 biomass, this aspect is not taken into account when standardized material is used in the
496 experiments.

497
498 The reviewer is of course correct to state that with eutrophication, also the quality of the
499 biomass produced in the system would change with potentially important consequences for the
500 decay process. Thus, interpretation of the results obtained with standardized litter need to be
501 conducted cautiously. In the discussion it now reads as follows:

502
503 “Standardized approaches like this, or also the cotton-strip assay (e.g. Latter and Walton, 1988), are useful
504 to separate the effects of environmental factors other than OM quality on decomposition processes and to
505 assess their relative importance. Otherwise, complex interaction effects of the abiotic environment and
506 OM quality make it difficult to predict the relevance of certain environmental factors for decomposition
507 processes, potentially masking the effects of important global-change drivers (Prescott, 2010). At the same
508 time, however, the global-change factors considered in the present study are likely to induce changes in
509 the quality of the OM accumulating in tidal wetlands, for instance through shifts in plant-species
510 composition and plant-tissue quality, that can potentially counterbalance or amplify the effects on
511 decomposition processes suggested here. Future research therefore needs to address OM quality feedbacks
512 on decomposition processes in tidal wetlands in order to gain a more complete understanding of global-
513 change effects on tidal-wetland stability and carbon-sequestration capacity.”

514
515 We agree with the reviewer that this sentence needs to be toned down in the abstract, because
516 there is no space for further elaboration on the assumptions and methodological
517 considerations. We will go with “potentially high sensitivity of OM stabilization to global change.”

518
519 L90-95: an important caveat here is that you only study the decomposition of one type of
520 source material (well, in two versions), but not other sources that contribute to the OM pool
521 e.g. marine or other aquatic inputs into the intertidal system.

522
523 We agree; this is important for the interpretation of our results. However, conventional litter
524 bag experiments are also restricted in their choice of material; here actually lies an advantage of
525 the standardized approach, although we acknowledge that the quality of the TBI materials is
526 obviously closer to that of wetland plant litter than to the marine derived, labile allochthonous
527 organic input a tidal wetland receives. We have elaborated on this in 4.4:

528
529 “Interpretation of results obtained from standardized approaches like the present needs to be made
530 cautiously because OM quality (i.e. its chemical composition) is a key parameter affecting its
531 decomposition. As the quality of the TBI materials differ from that of wetland plant litters, and likely even
532 more from the quality of the imported allochthonous OM (Khan et al., 2011), we did not expect to capture

533 actual rates of early-stage OM break-down in this study. Instead, we used the TBI to characterize the
534 decomposition environment by obtaining a measure for the potential to decompose and stabilize the
535 deployed standardized material.”

536

537 Additionally, we toned down our Implications (4.5):

538

539 “This study addresses the influence of temperature, relative sea level, and coastal eutrophication on the
540 initial transformation of biomass to SOM, and it does not encompass their effects on the existing SOM
541 pool. However, aspects of S and k are key components of many tidal wetland resiliency models (Schile et
542 al., 2014; Swanson et al., 2014) that have highlighted the critical role of the organic contribution to marsh
543 elevation gain. Although actual rates of S and k cannot be inferred from this study using a standardized
544 approach, our data identify strong negative effects of temperature, relative sea level, and coastal
545 eutrophication on the stabilization of fresh organic inputs to tidal-wetland soils. We argue that these
546 unanticipated combined effects yield the potential to strongly accelerate carbon turnover in tidal wetlands,
547 thus increasing their vulnerability to accelerated SLR, and we highlight the need for experimental studies
548 assessing the extent to which the here identified effects translate into native OM dynamics.”

549

550 section 2.2: it is important for the readers not familiar with the Keuskamp et al. paper to re-
551 iterate and stress the assumptions on which this approach is based, e.g. that k_2 (decomposition
552 constant of the non-labile fraction) is assumed to be 0, and that S is assumed to be similar for
553 both types of tea. I still find this major shortcomings- we know the first assumption not to be
554 valid, and I have not seen strong arguments to support the second assumption. The main reason
555 to make these assumptions is to allow to estimate k and S using only one time point of
556 measurements instead of having to measure at different points in time. These aspects deserve
557 to be mentioned explicitly and the limitations of the approach should be discussed more
558 critically. -What is the added value of this approach compared to simply measuring the decay of
559 the biomass over a limited number of time steps, and using a more realistic decay model?

560

561 Please note that Keuskamp et al. (2013) show that the TBI is quite robust against deviations
562 from the assumption that S is the same for the two materials. The assumption that k_2 could be
563 considerably higher than 0 during 3 months of deployment has already been questioned by us
564 (520-522). However, the reviewer brings important points here. In accordance, we elaborated
565 on the description of the TBI calculations in the respective section of the Methods (2.2):

566

567 “Decomposition rate (k) and stabilization (S) were assessed following the TBI protocol (Keuskamp et al.,
568 2013). The TBI approach can be considered as a simplified litter-bag approach, allowing a time- and cost-
569 efficient characterization of the decomposition environment, because k and S can be estimated without
570 repeated sampling of the decomposing material as in conventional approaches. This implies the
571 assumptions that (1) S is equal for the two types of material used in the approach and (2) that
572 decomposition of non-hydrolyzable materials during the 3 months of deployment is negligible. We refer
573 the reader to Keuskamp et al. (2013) for further detail and validity assessments of assumptions.”

574

575 The advantage of the TBI approach over a longer-term litter experiment is the time efficiency
576 that allowed us to assess decomposition in a large number of sites during the same growth
577 season and find enough collaborators capable to contribute with their work. Obviously, as
578 outlined by the inventors of the method (Keuskamp et al. 2013), the TBI can't substitute the
579 precision of classic litter bag methods, but it considerably reduces the effort necessary to

580 fingerprint local decomposition. It is a trait-off between precision and effort that helps
581 gathering decomposition data across ecosystems and biomes.

582
583 In order to demonstrate the usefulness of the method and its comparability to other methods
584 assessing decomposition processes tidal wetlands, we will separately present our data on k and
585 S for the sites along the North American East coast latitudinal gradient, along which previous
586 studies have shown clear temperature and latitudinal effects on decomposition processes. For
587 instance, Kirwan et al. (2014; Biogeosciences) demonstrated a strong increase in cellulose decay
588 with both temperature and latitude, and Mozdzer et al. (2014; Ecology) showed a marked
589 decrease in sulfate reduction with latitude along this transect. The TBI parameters assessed
590 along the same transect are in tight agreement with the previously reported results, particularly
591 the findings by Kirwan et al. (2014), demonstrating the usefulness of the method to characterize
592 the decomposition environment of tidal wetland soils.
593

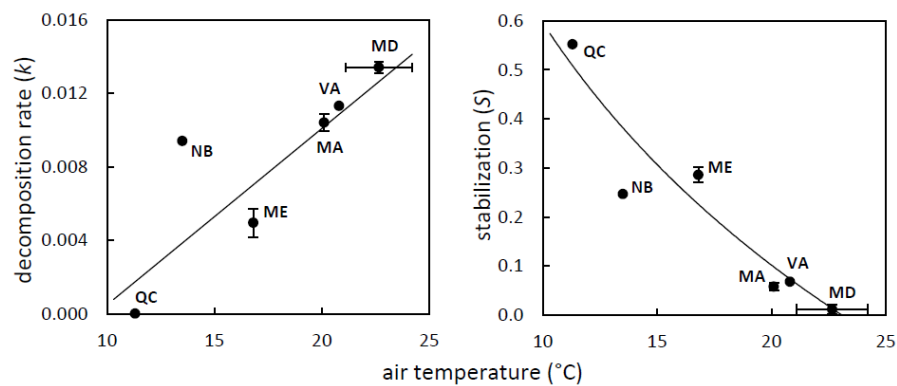


Figure S2 Site means of decomposition rate (left) and stabilization (right) versus mean air temperature of the deployment period shown for the ten sites situated along the latitudinal gradient of the North American Atlantic coast; state acronyms are shown (compare Table 1). Regression lines illustrate significant relationships[^]; regression models with lowest standard error of estimate (SEE) and highest R^2 are shown. Decomposition rate: $y = 0.001x - 0.0091$; $R^2 = 0.692$; $SEE = 0.003$; stabilization: $y = -0.712\ln(x) + 2.2331$; $R^2 = 0.860$; $SEE = 0.070$

594
595
596 Lastly, we discuss the applicability of the TBI approach in 4.4:

597
598 “Future research will have to test the applicability of the TBI approach in different ecosystems and test the
599 validity of its assumptions (i.e. S is equal for both types of material used, and mass loss of non-
600 hydrolyzable material is negligible over 3 months of deployment). The results of our regional scale
601 assessment along the North American Atlantic coast transect are in tight agreement with previously
602 reported results on cellulose break-down and soil microbial activity along this well studied transect
603 (Kirwan et al., 2014; Mozdzer et al., 2014). We can thereby demonstrate the usefulness of the TBI
604 approach to assess early-stage decomposition in tidal-wetland soils.”

605
606
607 L212-214: provide the data from Keuskamp et al. as well, we cannot compare or assess how
608 much higher your data are.

609
610 **Good idea, both are included now (Table S1).**

611
612 L 427-434: This is somewhat problematic also. It demonstrates the disadvantages of using these
613 operationally defined indices; to which extent is this caused by the assumption that S is identical
614 for the two types of substrate?

615
616 **We agree that the operational definition can cause problems and its implications have been**
617 **addressed as mentioned further up. The assumption that S is identical for the two types of**
618 **substrate is irrelevant in this context: S is determined only based on what is left of the green tea**
619 **material after incubation. The problem discussed in this section is that more material was**
620 **decomposed from the green tea material than theoretically possibly based on its hydrolysable**
621 **fraction. So either non-hydrolysable material was also decomposed to a considerable degree (as**
622 **also mentioned by the reviewer further up) or the hydrolysable fraction is in fact higher than**
623 **previously described.**

624
625 Secondly, keep in mind that anaerobic decomposition processes are important in tidal wetlands,
626 and can occur at high rates (similar order of magnitude as aerobic decomposition) up to
627 substantial depths.

628
629 **This is a valid point that needed more consideration. We specified in section 4.2:**

630
631 **“In tidal wetlands, differences in flooding frequency along elevational gradients often induce sharp**
632 **gradients in oxygen availability and redox conditions (Davy et al., 2011; Kirwan et al., 2013; Langley et**
633 **al., 2013), with potentially strong influence on OM decomposition and carbon cycling. However, the**
634 **effect of redox conditions on OM break-down is determined by the chemical quality of the decomposing**
635 **material: Decomposition of aged or recalcitrant OM can indeed be slower and incomplete in the absence**
636 **of oxygen, whereas the break-down of fresh and labile OM can be largely unaffected by oxygen**
637 **availability (Benner et al., 1984; Kristensen et al., 1995). Thus, also decomposition rate and stabilization**
638 **of labile, hydrolyzable OM, as assessed in the present study, is not necessarily affected by redox**
639 **conditions. Here, we demonstrate ...”**

640
641

642

643

644

645

646

647

648

649

650 **2. Marked-up manuscript version (text only)**

651

652

653 Global ~~change~~ effects on early-stage decomposition processes in tidal
654 wetlands: ~~implications~~ – Implications from a global survey using
655 standardized litter

656

657 **Authors**

658 Peter Mueller*¹, Lisa M. Schile-Beers², Thomas J. Mozdzer³, Gail L. Chmura⁴, Thomas Dinter⁵,
659 Yakov Kuzyakov^{5,6}, Alma V. de Groot⁷, Peter Esselink^{8,9}, Christian Smit⁹, Andrea D'Alpaos¹⁰,
660 Carles Ibáñez¹¹, Magdalena Lazarus¹², Urs Neumeier¹³, Beverly J. Johnson¹⁴, Andrew H.
661 Baldwin¹⁵, Stephanie A. Yarwood¹⁵, Diana I. Montemayor¹⁶, Zaichao Yang¹⁷, Jihua Wu¹⁷, Kai
662 Jensen¹, and Stefanie Nolte¹

663 **Affiliations**

- 664 1) Applied Plant Ecology, Biocenter Klein Flottbek, Universität Hamburg, Ohnhorststraße 18, 22609
665 Hamburg, Germany
- 666 2) Smithsonian Environmental Research Center, 647 Contees Wharf Rd, Edgewater, MD, 21037,
667 USA
- 668 3) ~~Bryn Mawr College~~, Department of Biology, Bryn Mawr College, 101 N. Merion Ave, Bryn
669 Mawr, PA, 19010, USA
- 670 4) Department of Geography, McGill University, 805 Sherbrooke St W, QC H3A 0B9, Canada
- 671 5) ~~Dept.~~Department of Soil Science of Temperate Ecosystems, University of Goettingen, Büsgenweg
672 2, 37077 Göttingen, Germany
- 673 6) Institute of Environmental Sciences, Kazan Federal University, 420049 Kazan, Russia
- 674 7) Wageningen Marine Research, Wageningen University & Research, Den Helder, Ankerpark 27,
675 1781AG, The Netherlands
- 676 8) ~~-~~PUCCIMAR, Boermarke 35, 9481 HD, Vries, The Netherlands
- 677 9) Conservation Ecology Group, Groningen Institute for Evolutionary Life Sciences, University of
678 Groningen, P.O. Box 11103, 9700 CC, Groningen, The Netherlands
- 679 10) Department of Geosciences, University of Padova, Via Gradenigo 6, Padua 35131 Italy
- 680 11) IRTA Aquatic Ecosystems, Carretera Poblenou Km 5.5, 43540 Sant Carles de Ràpita, Catalonia,
681 Spain
- 682 12) Department of Plant Taxonomy and Nature Conservation, University of Gdansk, ul. Wita Stwosza
683 59, 80-308 Gdansk, Poland
- 684 13) Institut des sciences de la mer de Rimouski, Université du Québec à Rimouski, -310 allée des
685 Ursulines, Rimouski QC G5L 3A1, Canada
- 686 14) Department of Geology, Bates College, 214 Carnegie Sciences Building, Lewiston, ME, 04240,
687 USA
- 688 15) Department of Environmental Science & Technology, University of Maryland, College Park, MD
689 20742, USA
- 690 16) Instituto de Investigaciones Marinas y Costeras (IIMyC), CONICET, UNMDP, Mar del Plata,
691 Argentina
- 692 17) Ministry of Education Key Laboratory for Biodiversity Science and Ecological Engineering,
693 Institute of Biodiversity Science, Fudan University, Shanghai 200438, PR China

694

695 *Corresponding author:

696 Peter Mueller; peter.mueller@uni-hamburg.de

697

698 Manuscript type: Primary research

699 **Abstract**

700 Tidal wetlands, such as tidal marshes and mangroves, are hotspots for carbon sequestration. The

701 preservation of organic matter (OM) is a critical process by which tidal wetlands exert influence

702 over the global carbon cycle and at the same time gain elevation to keep pace with sea-level rise

703 (SLR). The present study ~~provides~~assessed the ~~first global-scale field-based experimental~~

704 ~~evidence effects~~ of temperature and relative sea level ~~effects~~ on the decomposition rate and

705 stabilization of OM in tidal wetlands. ~~The study was conducted in 26 marsh and mangrove sites~~

706 ~~across four continents worldwide~~, utilizing commercially available standardized OMlitter. While

707 effects on decomposition rate per se were minor, we show ~~unanticipated and combined~~strong

708 negative effects of temperature and relative sea level on ~~OM~~stabilization, as based on the

709 fraction of labile, rapidly hydrolyzable OM that becomes stabilized during deployment. Across

710 study sites, OM stabilization was 29% lower in low, more frequently flooded vs. high, less

711 frequently flooded zones. ~~OM stabilization~~Stabilization declined by ~~~90~~75% over the studied

712 temperature gradient from 10.9 to 28.5°C, ~~corresponding to a decline of ~5% over a 1°C~~

713 ~~temperature increase~~. Additionally, data from the Plum Island long-term ecological research site

714 in Massachusetts, US show a pronounced reduction in OM stabilization by >70% in response to

715 simulated coastal eutrophication, confirming the potentially high sensitivity of OM stabilization

716 to global change. We therefore provide evidence that rising temperature, accelerated SLR, and

717 coastal eutrophication may decrease the future capacity of tidal wetlands to sequester carbon by

718 affecting the initial transformations of recent OM inputs to soil ~~organic matter~~OM.

719

720

721 1 Introduction

722 Tidal wetlands, such as marshes and mangroves, provide a wide array of ecosystem services that
723 have been valued at approximately US\$ 10,000 per hectare and year, making them some of the
724 most economically valuable ecosystems on Earth (Barbier et al., 2011; Kirwan and Megonigal,
725 2013). Yet, tidal wetlands are threatened and vulnerable ecosystems, experiencing pronounced
726 loss through global-change impacts, such as land use (Pendleton et al., 2012) and accelerated
727 sea-level rise (SLR) (Craft et al., 2009; Crosby et al., 2016). In recent years, carbon
728 sequestration has increasingly been recognized as an ecosystem service of tidal wetlands (e.g.
729 Chmura et al., 2003; Mcleod et al., 2011). ~~Tidal wetlands are efficient long-term carbon sinks,~~
730 ~~preserving organic matter (OM) for centuries to millennia.~~ Here, high rates of OM input (from
731 both autochthonous and allochthonous production) co-occur with reducing soil conditions and
732 thus slow rates of decomposition, leading to ~~organic~~long-term carbon ~~sequestration~~ rates that
733 exceed those of most other ecosystem types by ~~orders~~an order of magnitude (Mcleod et al.,
734 2011). At the same time, suppressed decomposition and the preservation of OM is a primary
735 process by which many tidal wetlands gain elevation and keep pace with rising sea level (Kirwan
736 and Megonigal, 2013). Consequently, global changes that ~~might increase~~decrease OM
737 ~~decomposition~~preservation in tidal ~~wetland~~ soils not only affect carbon sequestration, but also
738 decrease ecosystem stability against SLR. It is therefore critical to identify global ~~change~~ factors
739 that affect the transformation of organic inputs to stable soil OM (SOM) in tidal wetlands and to
740 assess the magnitude of their effects.

741 There are multiple methods for assessing factors that influence carbon sequestration,
742 including direct measurements of plant production, ~~organic~~ carbon stocks, accretion, and
743 decomposition rates. Litter-bag techniques assessing the weight loss of plant material over time
744 are probably the easiest way to measure decomposition rates in situ and have been widely used

745 since the 1960s (Prescott, 2010). Global-scale assessments of litter decomposition have been
746 conducted as both meta-analyses (e.g. Zhang et al., 2008) and as inter-site studies along
747 latitudinal gradients (Berg et al., 1993; ~~Trofymow et al., 2002; McTiernan et al., 2003;~~
748 Cornelissen et al., 2007; McTiernan et al., 2003; Powers et al., 2009; Trofymow et al., 2002) in
749 order to assess effects of climate parameters, mostly with focus on temperature and moisture
750 gradients, on decomposition rate. Besides abiotic or climate effects, these studies could also
751 identify litter quality itself as an important predictor for decomposition rate (Zhang et al., 2008).

752 Relationships between single climate or litter-quality parameters and decomposition rate
753 often are not linear. Instead, complex interactions between litter-quality and climate parameters
754 seem to control litter decomposition (Zhang et al., 2008), creating challenges in separating
755 climate from litter-quality effects and predicting the relevance of potential global-change drivers
756 for decomposition rate. In order to separately assess environmental or climate effects on litter
757 decomposition at a global scale, it is therefore necessary to standardize litter quality in inter-site
758 studies. However, implications of litter-decay data for carbon sequestration need to be considered
759 cautiously, as the link among litter-decomposition rate, SOM formation, and ultimately carbon
760 sequestration is not straightforward (~~Prescott, 2010~~; Cotrufo et al., 2013; Prescott, 2010):
761 Because plant tissues are not resistant to decay per se, it is critical to understand their
762 biogeochemical transformation into stable compounds that leads to the formation of SOM (i.e.
763 stabilization) rather than understanding the pace at which early ~~stages of stage~~ decomposition
764 proceeds (~~Prescott, 2010~~; Castellano et al., 2015; Haddix et al., 2016; Prescott, 2010).

765 Keuskamp and others (2013) developed an efficient approach for studying litter
766 decomposition and OM transformation at a global scale, using commercially available tea as
767 standardized material. Their Tea Bag Index (TBI) approach is based on the deployment of two
768 types of tea that considerably differ in their OM quality. The method allows for the determination

769 of the decomposition rate constant (in the following referred to as decomposition rate or k), ~~as in~~
770 ~~classic litter bag approaches,~~) and a stabilization factor (in the following referred to as
771 stabilization or S), which describes the fraction of labile and rapidly decomposable OM that
772 becomes stabilized during deployment. _____

773 In the present study, we assessed effects of the ~~impacts of multiple~~ global-change factors
774 ~~—global~~ warming, ~~sea-level rise (accelerated SLR), salt-water intrusion,~~ and coastal
775 eutrophication —on both OM decomposition rate and stabilization in tidal—wetland soils by
776 conducting a worldwide field study survey using standardized litter. First, by covering a large
777 temperature gradient of $\Delta T > 15$ °C across sites, we aimed to capture temperature effects on OM
778 decomposition rate and stabilization, thereby improving our ~~process-level~~ understanding on how
779 global warming affects carbon turnover and ultimately sequestration in tidal wetlands. Second, by
780 conducting paired measurements in both high- and low-elevated zones of tidal wetlands
781 worldwide, we were aiming to gain insight into potential effects of accelerated ~~relative~~ SLR on
782 carbon turnover. Despite the dominant paradigm that decomposition is inversely related to
783 flooding, the existing literature on hydrology and SLR effects on OM decomposition in tidal
784 wetlands yields equivocal results, which is often due to the overriding effect of OM quality on
785 decomposition rate (Hemminga and Buth, 1991; Kirwan et al., 2013; Mueller et al., 2016).

786 ~~Additionally, by expanding our study to include fresh and brackish sites, we anticipated to~~
787 ~~capture the effects of salt-water intrusion into brackish and fresh systems, which is likely to affect~~
788 ~~decomposition processes in tidal wetlands (Morrissey et al., 2014). Specifically, high~~
789 ~~concentrations of dissolved sulfate in seawater, acting as an alternative terminal electron~~
790 ~~acceptor, can enhance anaerobic microbial metabolism in systems with lower salinity (Meronigal~~
791 ~~et al., 2004; Sutton-Grier et al., 2011). Lastly, we used the long-term ecological research site of~~
792 ~~the TIDE project~~ Lastly, we used the TIDE project plots of the Plum Island long-term ecological

793 | [research site](#) in Massachusetts, US (Deegan et al., 2012) to experimentally assess both the effects
794 of coastal eutrophication and – with respect to SLR-driven increases in flooding frequency – the
795 relevance of nutrient delivery through floodwater for the early stages of OM decomposition in
796 tidal wetlands.

797

798

799

800

801

802

803

804

805

806

807

808

809

810

811

812 2 Methods

813 2.1 Study sites and experimental design

814 The ~~study~~worldwide survey was conducted ~~in 26 tidal wetlands~~ during the 2015 growing season
815 ~~(Figure and included a total of 30 tidal-wetland sites. Sites were partly co-located within larger~~
816 ~~coastal and estuarine regions (Fig. 1, Table 1). ~~Nine~~Eleven sites were situated along the European~~
817 ~~coasts of the North Sea, Mediterranean, and Baltic, ~~ten~~Thirteen sites were located along the East~~
818 ~~and West coasts of North America including the St. Lawrence Estuary, Bay of Fundy,~~
819 ~~Chesapeake Bay, and San Francisco Bay, and four mangrove sites were situated along the~~
820 ~~Caribbean coast of Central America in Belize and Panama. Additionally, one Chinese site~~
821 ~~(Yangtze Estuary) and one Argentinian site were included in our study. Sixteen of the sites were~~
822 ~~salt marshes, ~~six~~ten were tidal freshwater and brackish sites, and four sites were mangroves. ~~At~~~~
823 ~~22In 21 sites, we compared high and low elevated zones, which were characterized by distinct~~
824 ~~plant species compositions (i.e. different communities in high vs. mid vs. low marshes) or by~~
825 ~~different stature of mangroves (i.e. dwarf vs. fringe phenotypes). We used relative elevation (i.e.~~
826 ~~high vs. low elevated zone) as a site-specific proxy for relative sea level. By doing so, we did not~~
827 ~~capture the actual variability in the tidal inundation regime across our study sites as these vary in~~
828 ~~absolute elevation and in elevation relative to mean high water. Finally, we included the long-~~
829 ~~term experimental site of the TIDE project in Massachusetts, US to assess effects of nutrient~~
830 ~~enrichment on litter-decomposition rate and stabilization. Through nitrate additions to the~~
831 ~~incoming tides on at least 120 days per year, nutrient enriched areas at the TIDE project site~~
832 ~~receive floodwater with 10-15 fold increased nitrogen (N) concentrations compared to reference~~
833 ~~areas since 2004. From 2004-2010 also phosphate was added to the floodwater; however, this~~
834 ~~has been discontinued because creek water P concentrations are high enough to prevent~~

835 secondary P limitation through N enrichment (details in Deegan et al., 2012; Johnson et al.,
836 2016).

837 Decomposition rate and stabilization were measured by deploying tea bags in ten points
838 per zone or treatment within a site (n=10). Spacing between replicates within a zone or treatment
839 was ≥ 2 m. However, as sites differed considerably in their areal extent, the distribution and thus
840 spacing between points had to be adjusted to be representative for the given system. ~~Temperature~~
841 Air temperature for the period of deployment was measured at site, or temperature data was
842 obtained from the online service of ~~Aeeuweather~~AccuWeather (accuweather.com; accessed
843 12/25/2016) for locations within a distance of 15 km to the site for most sites, but not further than
844 60 km for some remote sites. It needs to be noted here, that top-soil temperature would differ
845 from air temperature depending on factors such as canopy shading or tidal regime and water
846 temperature. As a consequence, air temperature can only approximate the temperature conditions
847 of the actual decomposition environment (Fig. S3).

848 2.2 Decomposition rate and stabilization measurements

849 Decomposition rate (k) and stabilization ~~factor~~(S) were assessed following the ~~Tea-Bag~~
850 ~~Index~~TBI protocol (Keuskamp et al., 2013). ~~Briefly, at~~The TBI approach can be considered as a
851 simplified litter-bag approach, allowing a time- and cost-efficient characterization of the
852 decomposition environment, because k and S can be estimated without repeated sampling of the
853 decomposing material as in conventional approaches. This implies the assumptions that (1) S is
854 equal for the two types of material used in the approach and (2) that decomposition of non-
855 hydrolyzable materials during the 3 months of deployment is negligible. We refer the reader to
856 Keuskamp et al. (2013) for further detail and validity assessments of assumptions.

857 At each point of measurement, two nylon tea bags (200 µm mesh size), one containing
858 green tea (EAN: 8 722700 055525; Lipton, Unilever + ~~PepsiCo~~, UK) and one containing rooibos
859 (8 722700 188438, Lipton, Unilever + ~~PepsiCo~~, UK), were deployed as pairs in ~8 cm soil depth,
860 separated by ~5 cm. The initial weight of the contents was determined by subtracting the mean
861 weight of ~~10~~ten empty bags (bag + string + label) from the weight of the intact tea bag prior to
862 deployment (content + bag + string + label). The tea bags were retrieved after an incubation time
863 of ~~~90~~ 92 ± 6 (SD) days, with three sites having an incubation period >100 days and one site <80
864 days. Upon retrieval, tea bags were opened, and tea materials were carefully separated from ~~clay~~
865 ~~particles and~~ fine roots and soil, dried for 48 h at 70°C, and weighed.

866 Calculations for k and S followed Keuskamp et al. (2013):

867

868 Eq 1)
$$W_r(t) = ar e^{-kt} + (1-ar)$$

869

870 Eq 2)
$$S = 1 - ag / H_g$$

871

872 Eq 3)
$$ar = H_r (1-S)$$

873

874 $W_r(t)$ describes the substrate weight of rooibos after incubation time (t in days), ar the labile and
875 1-ar the recalcitrant fraction of the substrate, and k is the decomposition rate constant. S describes
876 the stabilization factor, ag the decomposable fraction of green tea (based on the mass loss during
877 incubation) and H_g the ~~hydrolysable~~hydrolyzable fraction of green tea. The decomposable

878 | fraction of rooibos tea is calculated in Eq 3 based on its ~~hydrolysable~~hydrolyzable fraction (Hr)
879 | and the stabilization factor S. With $W_r(t)$ and a_r known, k is calculated using Eq 1.

880 | In accordance with Keuskamp et al. (2013), extractions for determination of the
881 | ~~hydrolysable~~hydrolyzable fractions of green and rooibos tea followed Ryan et al. (1990).
882 | However, instead of using Ryan's forest products protocol we conducted the alternative forage
883 | fiber protocol for the determination of the ~~hydrolysable~~hydrolyzable fraction. Briefly, 1 g of dried
884 | tea material (70°C for 24 h) was boiled in cetyltrimethyl ammonium bromide (CTAB) solution (1
885 | g CTAB in 100 ml 0.5 M H₂SO₄) for 1 h (~~Ryan et al., 1990~~; Brinkmann et al., 2002; Ryan et al.,
886 | 1990). The extract was filtered through a 16-40- μ m sinter filter crucible (Duran, Wertheim,
887 | Germany) using a water-jet vacuum pump and washed with 150 ml of hot water followed by
888 | addition of acetone until no further de-coloration occurred (Brinkmann et al., 2002). The
889 | remaining material was left in the sinter, dried for 12 h at 70°C, cooled in a desiccator and
890 | weighed. 20 mL of 72% H₂SO₄ was added to the sinter and filtered off after an incubation of 3 h,
891 | followed by washing with hot water to remove remaining acid. The sinter was dried at 70°C for
892 | 12 h, cooled in a desiccator, and weighed to determine the non-~~hydrolysable~~hydrolyzable
893 | fraction. Finally, the sinter containing the remaining sample was ignited at 450°C for 3 h in order
894 | to determine the ash content of the material.

895 | In addition to the determination of the ~~hydrolysable~~hydrolyzable fraction, we measured
896 | total C and N contents of the tea material using an elemental analyzer (HEKAtech, Wegberg,
897 | Germany). The ~~hydrolysable~~hydrolyzable fraction of both green and rooibos tea was higher than
898 | reported in Keuskamp et al. (2013) (Table 2). However, the determined C and N contents of the
899 | tea materials are in agreement with those reported in Keuskamp et al. (2013) (Table 2).
900 | Therefore, deviations from the ~~hydrolysable~~hydrolyzable fraction as reported previously are

901 likely due to the less conservative extraction assessment in the present study and not due to actual
902 changes in the quality of the materials.

903 2.3 Data ~~Analyses~~mining

904 ~~For all across-site analyses, mean values of each site by elevation zone (or site by salinity class)~~
905 ~~combination were used (N=51).~~ Relationships between single parameters and litter
906 decomposition are often not linear. Instead, critical thresholds seem to exist at which a certain
907 predictor (e.g. mean annual temperature) becomes influential (~~Rothwell et al., 2008; Prescott,~~
908 ~~2010~~Prescott, 2010; Rothwell et al., 2008). In the first step of data mining, we therefore used
909 classification and regression tree analysis (CART) to identify potential thresholds and important
910 predictors for k and S (Fig. S1). Data mining was conducted using STATISTICA 10 (StatSoft
911 Inc., Tulsa, OK, USA).

912 ~~———— In the first step of our data analysis, we therefore used classification and regression tree~~
913 ~~analysis (CRTA) to identify important predictors for k and S. CRTA is a non-parametric~~
914 ~~procedure for the step-wise splitting of the data set with any number of continuous or categorical~~
915 ~~and correlated or uncorrelated predictor variables (Breiman et al., 1984; Rothwell et al., 2008),~~
916 ~~and it has been recommended to identify thresholds and to handle large-scale decomposition data~~
917 ~~sets (Rothwell et al., 2008; Prescott, 2010). We conducted CRTA separately for k and S using~~
918 ~~temperature, salinity class, tidal amplitude, ecosystem type, soil type, and relative elevation as~~
919 ~~predictor variables (Table 1). V-fold cross validation was set at 5 (as commonly conducted,~~
920 ~~compare Rothwell et al. (2008)), and the minimum number for observations per child node was~~
921 ~~set at n = 4, corresponding to at least two sites or 8% of the total data set.~~

922 2.4 Statistical analyses

923 To test for ~~correlations between the variables salinity class,~~effects of temperature on k and S,
924 Spearman rank correlations were conducted using site means (n = 30). As we did not expect
925 temperature to be independent of other parameters in this observational study, we constructed a
926 Spearman correlation matrix including the parameters temperature, latitude, tidal amplitude, k
927 ~~and S, Spearman rank correlations were used (Table 3). Mann-Whitney U tests were conducted to~~
928 ~~test for salinity class, k, and S. Additionally, we tested for~~ differences in ~~k and S~~these parameters
929 between marshes and mangroves and betweensites with mineral and organic ~~soil types.~~

930 ~~We tested for linear effects of temperature on k and S across sites~~soils, using simple linear
931 regression analyses (Fig. 2). Two-tailed paired t-tests were used toMann-Whitney U tests (Table
932 3). Curve fitting was used to further explore relationships between temperature, k, and S, and
933 regression models with lowest standard error of estimate and highest R² are displayed in Figure 2
934 and S₂.

935 To test for effects of relative elevation (as proxy for relative sea level) on k and S (Fig. 3):
936 ~~Subsequent one, two-tailed paired t-tests were conducted to test for.~~ Mean values of high and low
937 elevated zones of the same effect within mineral, organic, marsh, and mangrove systems
938 separately.

939 ~~In 21 of our 22~~ sites where tea bags were deployed in both high and low elevation zones,
940 ~~replication was sufficient to conduct~~ were used (n = 21). The absence of outliers and normal
941 distribution of the difference in the independent variable (as assessed visually) assured robustness
942 of paired t-tests. To assess the consistency of potential effects of relative elevation on k and S,
943 one-way ANOVA to test for differences in k and S between zones forwas used in each site
944 separately (Fig. S2).~~replication was sufficient in 20 sites). Normal distribution of residuals was~~
945 assessed visually, Levene's test was used to test for homogeneity of variance, and data were log-
946 transformed if assumptions were not met. Mann-Whitney U tests were conducted as a non-

947 parametric alternative when log-transformed data did not meet ANOVA assumptions (Table S1).

948 We tested for effects of nutrient enrichment on k and S in the data from the TIDE project
949 site (Massachusetts, US) using two-way ANOVA with enrichment treatment and marsh zone as
950 predictors. ~~All analyses were conducted using STATISTICA 10 (StatSoft Inc., Tulsa, OK,~~
951 ~~USA).~~ When Levene's test indicated heterogeneous variance (true for k), data were log-
952 transformed, which stabilized variance. Normal distribution of residuals was assessed visually.

953 Lastly, in order to assess the applicability of the TBI approach in tidal wetlands, we
954 separately investigated the temperature response of k and S for the ten sites situated along the
955 North American Atlantic coast (Fig. S2). Previous studies have shown clear
956 temperature/latitudinal effects on decomposition and microbial activity along this well-studied
957 transect (Kirwan et al. 2014; Mozdzer et al. 2014), allowing us to compare the TBI approach with
958 other methods. Regional-scale transects with sufficient temperature/latitudinal range along other
959 coastlines could not be identified (Fig. 1; Table 1). Statistical analyses were conducted using
960 STATISTICA 10 (StatSoft Inc., Tulsa, OK, USA).

961

962

963

964

965

966

967

968 3 Results

969 3.1 Temperature effects

970 We found no ~~linear (Fig. 2a) or monotonic (Table 3) relationships~~relationship between
971 temperature and ~~k~~across study sites (Fig. 2a; Table 3). Also, ~~CRTACART~~ revealed temperature
972 only as a ~~minor predictor~~subordinate splitting variable for k (~~Figure~~Fig. S1a). Specifically,
973 temperature seems to positively affect k in meso-tidal systems only (amplitude >2.1m; Fig. S1a;
974 node 5) with sites $\geq 14.5^{\circ}\text{C}$ during deployment supporting ~~considerably~~ higher rates of
975 decomposition than sites characterized by lower temperatures. However, this apparent
976 temperature effect was inconsistent within the group of observations with tidal amplitude >2.1m
977 (Fig. S1a; nodes 13-15). In contrast to the results of the global-scale assessment, k is strongly and
978 positively related with temperature across the ten sites situated along the North American
979 Atlantic coast, with temperature explaining approx. 70% of variability in k (Fig. S2). ~~Furthermore,~~
980 ~~the majority of sites (65%) are characterized by tidal amplitudes <2.1 m and show no temperature~~
981 ~~effect on k.~~

982 ~~In contrast to the temperature response of k, S~~Stabilization was strongly affected by
983 temperature (Fig. 2b; Table 3). The significant negative correlation between S and temperature (p
984 < 0.01 ; $r^2 = 0.287$; Fig. 2b; Table 3) agrees well with the ~~CRTACART~~ (Fig. S1b). However,
985 ~~CRTACART~~ also identified a narrow temperature range (21.9-23.6°C) in which increasing
986 temperature led to higher stabilization (Fig. S1b; node 11). This group of observations ($n = 5$)
987 from the general pattern is also clearly visible in Figure 2b and represents the 4 Mediterranean
988 sites (Ebro Delta and Venice Lagoon) of our survey. The positive relationship between
989 temperature and S was even clearer when focusing on the ten sites along the North American
990 Atlantic coast, with temperature explaining >85% of variability in S (Fig. 2b, S2).

991 Temperature was highly correlated with latitude and tidal amplitude, and temperature was
992 not independent of soil type (mineral/organic) and ecosystem type (marsh/mangrove) (Table 3).
993 The effect of latitude was similarly pronounced as the temperature effect on S – and consequently
994 –effects of these two parameters on S cannot be separated (Table 3). By contrast, tidal amplitude
995 and soil type did not significantly affect S, and the difference in S between mangroves and
996 marshes is only marginally significant (Table 3). These findings suggest that the presented
997 temperature effect on S occurs to be mainly independent of tidal amplitude and soil type.

998 3.2 Effects of relative sea level and nutrient enrichment

999 Paired comparisons of high vs. low elevated zones indicate no consistent effect of relative sea
1000 level on k across sites ($p > 0.1$; Fig. 3a+e), whereas S was significantly reduced by 29% in low
1001 compared to high elevated zones ($p < 0.01$; Fig. 3b). ~~This significant reduction of S in low vs.~~
1002 ~~high elevated zones was consistent across mineral and organic, as well as marsh and mangrove~~
1003 ~~systems (Fig. 3d).~~ Testing for effects of relative sea level within each site separately revealed that
1004 S is significantly reduced by 28-87% in the lower elevated zone in ~~1514~~ of ~~2120~~ sites. ~~A;~~
1005 ~~whereas a~~ significant increase of S in low vs. high elevated zones was found in none of ~~these~~
1006 ~~24~~ the 20 sites (Fig. ~~Table S1~~). This finding demonstrates the consistency of the sea-level effect
1007 on S irrespective of ecosystem type (marsh/mangrove) soil type (mineral/organic) and site
1008 salinity (brackish/salt). ~~S2~~ In ~~ten~~ nine of the sites, we also found a significant effect of relative
1009 sea level on k; ~~with~~. However, in six sites k was significantly higher ~~k~~ in low vs. high zones, and
1010 in ~~seven~~ three sites ~~and k was~~ significantly lower ~~k~~ in low vs. high zones ~~in three sites (Fig. S2).~~
1011 The direction of effects on k seems to be independent of ecosystem type, soil type, and site
1012 salinity (Table S1).

1013 ~~3.3~~ Effects of salinity and nutrient enrichment

1014 ~~We found no significant relationship between salinity class and k or S (Table 3). Also, CART did~~
1015 ~~not reveal salinity class as an important factor for k and S (Fig. S1), and no consistent salinity~~
1016 ~~effect on k and S was found when comparing sites of different salinities within single estuarine~~
1017 ~~regions (Chesapeake, Ebro Delta, Long Marsh, San Francisco Bay; Fig. S3).~~

1018 ——— The nutrient enrichment treatment at the TIDE project site decreased S by 72% in the high
1019 marsh (Fig. 3d). S in the low marsh likewise was similarly low as in the ~~fertilized~~enriched high
1020 marsh and not further reduced by ~~fertilization~~nutrient enrichment (Fig. 43d). In contrast, k was
1021 not responsive to the ~~fertilization~~nutrient enrichment treatment in neither low nor high marsh
1022 (Fig. 43c).

1023 3.4 Other factors influencing decomposition rate and stabilization

1024 ~~CRTA~~CART revealed tidal amplitude as an important predictor for k (Fig. S1a). However, this
1025 result needs to be ~~interpreted~~considered cautiously because splits based on tidal amplitude
1026 suggest mixed effects (Fig. S1a). Accordingly, no linear ($p > 0.68$; $r^2 = 0.004$) or
1027 monotoniesignificant relationship (Table 3) existed between tidal amplitude and k, ~~and effects of~~
1028 ~~tidal amplitude are not independent from other factors because strong correlations existed with~~
1029 ~~ecosystem and soil type, temperature, and latitude (Table 3).~~ across sites (Table 3). Soil type
1030 (mineral ~~vs.~~ /organic) and ecosystem type (marsh ~~vs.~~ /mangrove) did not affect k and S across
1031 sites (Table 3). We found no significant relationship between salinity class and k or S (Table 3;
1032 Fig.). Also, CART did not reveal salinity class ~~S1a).~~ In comparison, S was lower in mangroves
1033 than in marshes and lower in organic than in mineral systems (Table 3), presumably caused by
1034 temperature effects because ecosystem and soil type did not show up as predictors in CRTA an
1035 important factor for k and S (Fig. ~~S1b~~S1).

1036

1037 4 Discussion

1038 ~~The findings of the present study cannot demonstrate consistent effects of either temperature or~~
1039 ~~relative sea level on the decomposition rate of recent OM inputs (commonly assessed as k in litter~~
1040 ~~bag studies) in tidal wetlands. With respect to C sequestration, however, litter decay data need to~~
1041 ~~be considered cautiously, as the link among decomposition rate, SOM formation, and ultimately~~
1042 ~~C sequestration is not straightforward. That is, plant tissues and other fresh OM inputs into an~~
1043 ~~ecosystem are not resistant to decay per se, and as a consequence, it is critical to understand their~~
1044 ~~biogeochemical transformation into stable compounds that leads to the formation of SOM (i.e.~~
1045 ~~stabilization) rather than understanding the pace at which decomposition proceeds (Prescott,~~
1046 ~~2010; Castellano et al., 2015; Haddix et al., 2016). Here, we also assessed OM stabilization, and~~
1047 ~~in contrast to decomposition rate, stabilization decreased with temperature and was consistently~~
1048 ~~lower in low vs. high elevated zones of tidal wetlands. Our study therefore provides indirect~~
1049 ~~evidence that rising temperature and accelerated SLR could decrease the capacity of tidal~~
1050 ~~wetlands to sequester C by affecting the initial transformations of recent OM inputs to SOM (i.e.~~
1051 ~~stabilization).~~

1052 4.1 Temperature effects on decomposition processes

1053 ~~Surprising~~A positive relationship between temperature and decomposition rate was found only at
1054 the regional scale across the ten sites along the North American Atlantic coast (Fig. S2), but not
1055 across all sites at the global scale (Fig. 2). Even though this finding occurs surprising in the
1056 context of basic biokinetic theory, the temperature response of decomposition rate was weak or
1057 not it is in agreement with findings of Djukic and others (2018), demonstrating climate effects on
1058 the break-down of the TBI materials across terrestrial ecosystems at the biome scale, but not at
1059 the global scale across biomes.

1060 ~~The present. Following study used air temperature as a proxy for top-soil temperature.~~
1061 ~~Thus, the temperature regime of the decomposition environment was only approximated, which~~
1062 ~~certainly would have weakened a significant relationship between temperature and k. However,~~
1063 ~~following typical Q10 values for biological systems of 2-3 (Davidson & Janssens, 2006), k~~
1064 ~~should have at least doubled over the gradient of $\Delta T > 15^\circ\text{C}$. However, findings from: yet our~~
1065 ~~data do not even show a tendency of an effect ($r_s = 0.02$; Table 2). We therefore propose that~~
1066 ~~other parameters exerted overriding influence on k, mainly masking temperature effects, and~~
1067 ~~have not been captured by our experimental design. This notion is in line with the fact that~~
1068 ~~studies conducted at single-marsh to regional scales are not conclusive either report equivocal~~
1069 ~~results on the temperature response of k, ranging from no or small/moderate (Charles &~~
1070 ~~Dukes, 2009; Kirwan et al., 2014; Janousek et al., 2017; Kirwan et al., 2014) to strong~~
1071 ~~seasonally-driven temperature effects with a $Q_{10} > 3.4$ as found within a single site (Kirwan &~~
1072 ~~Blum, 2011). Although temperature sensitivity of OM types is variable (Craine et al., 2010;~~
1073 ~~Hines et al., 2014; Wilson et al., 2016), temperature sensitivity of the used TBI materials was~~
1074 ~~sufficiently demonstrated (Keuskamp et al., 2013). We therefore conclude that other parameters~~
1075 ~~exerted overriding influence on k, mainly masking temperature effects and have not been~~
1076 ~~captured by our experimental design. For instance, we do not have data on plant biomass~~
1077 ~~parameters that are thought to exert strong control on decomposition in tidal wetlands through~~
1078 ~~priming effects (Wolf et al., 2007; Mueller et al., 2016; Bernal et al., 2017). Likewise and Blum,~~
1079 ~~2011). For instance, large differences in site elevation and hydrology could have induced high~~
1080 ~~variability in k across sites and masked potential temperature effects. Indeed, we demonstrate~~
1081 ~~significant but mixed effects of relative sea level on k for some sites; (Table S2); however, we do~~
1082 ~~not have sufficient data on actual site elevation or hydrology to control for these factors as~~
1083 ~~covariates affecting the temperature effect on k. Likewise, we do not have data on nutrient~~

1084 availability, plant productivity, or various anthropogenic impacts that could have exerted strong
1085 control over decomposition processes in the studied sites (Deegan et al., 2012; Keuskamp et al.,
1086 2015a; Macreadie et al., 2017; Mueller et al., 2016).

1087 In contrast to ~~the missing or subtle effect~~effect of temperature on k, OM stabilization was
1088 strongly affected ~~by temperature.~~ Overall, S decreased by ~~~9075%~~ over our temperature gradient
1089 from 10.9 to 28.5°C, ~~corresponding to a decline of ~5% over a 1°C temperature increase (Figure~~
1090 (Fig. 2b). Thus, we demonstrate a considerable temperature effect on the initial steps of biomass
1091 decomposition in tidal wetlands. ~~This effect, however,~~However, as also demonstrated for k, the
1092 temperature effect on S was much clearer at the regional scale when focusing on the sites along
1093 the North American Atlantic coast (Fig. S2), suggesting high variability in S across regions
1094 irrespective of the temperature regime. In accordance, we also demonstrate a clear divergence of
1095 the four Mediterranean sites from the regression model (Fig. 2; S1), which could be related to
1096 differences in precipitation or nutrient availability across study regions. Future experimental
1097 work is therefore required in order to further assess the effects of temperature on OM
1098 stabilization and to separate temperature from latitudinal and other interacting effects (e.g. as
1099 outlined above for k) that are difficult to control for in observational studies.

1100 The temperature effect on the initial steps of biomass decomposition we identified in the
1101 present study is not driven by changes in decomposition rate per se, but – more importantly – by
1102 affecting the transformation of fresh and rapidly decomposable organic matter into stable
1103 compounds, with potentially important implications for Ccarbon sequestration (e.g. Cotrufo et al.,
1104 2013).

1105 In their global-scale assessment, Chmura et al. (2003) indeed report a negative
1106 relationship of soil organic C density and mean annual temperature within both salt marshes and
1107 mangroves. Indeed, Chmura and colleagues hypothesized stimulated microbial decomposition at

1108 higher temperatures to be the responsible driver ~~for~~of this relationship. Plant production and thus
1109 OM input is known to increase with latitude and temperature in tidal wetlands (Baldwin et al.,
1110 2014; Charles & Dukes, 2009; Gedan & Bertness, 2009; Kirwan et al., 2009; Baldwin et
1111 al., 2014), but this increase seems to be more than compensated by higher microbial
1112 decomposition. Working ~~on~~at the same spatial scale as Chmura et al. (2003), our study strongly
1113 supports this hypothesis and provides ~~the~~ mechanistic insight into the temperature control of OM
1114 decomposition as an important potential driver of Carbon sequestration in tidal wetlands.

1115 4.2 Relative-sea-level effects on decomposition processes

1116 Flooding and thus progressively lower oxygen availability in soil is supposed to be a strong
1117 suppressor of decomposition (Davidson and Janssens, 2006). In tidal wetlands, differences in
1118 flooding frequency along elevational gradients often induce sharp gradients in oxygen availability
1119 and redox conditions (Davy et al., 2011; Kirwan et al., 2013; Langley et al., & Janssens, 2006).
1120 Despite this dominant paradigm, we clearly demonstrate that k is not reduced in low vs. high
1121 elevated zones of tidal wetlands (Fig. 2013), with potentially strong influence on OM
1122 decomposition and carbon cycling. However, the effect of redox conditions on OM break-down
1123 is determined by the chemical quality of the decomposing material: Decomposition of aged or
1124 recalcitrant OM can indeed be slower and incomplete in the absence of oxygen, whereas the
1125 break-down of fresh and labile OM can be largely unaffected by oxygen availability (Benner et
1126 al., 1984; Kristensen et al., 1995). Thus, also decomposition rate and stabilization of labile,
1127 hydrolyzable OM, as assessed in the present study, is not necessarily affected by redox
1128 conditions. Here, we demonstrate that k is not reduced in low (more frequently flooded) vs. high
1129 elevated (less frequently flooded) zones of tidal wetlands (Fig. 3a). This finding is in accordance
1130 with an increasing number of studies demonstrating negligible direct effects of sea level on

1131 decomposition rate in tidal wetland soils ([Janousek et al., 2017](#); Kirwan et al., 2013; Mueller et
1132 al., 2016; ~~Janousek et al., 2017~~). ~~A SLR induced reduction in decomposition rate with positive~~
1133 ~~feedback on tidal wetland stability seems therefore to be an unlikely scenario.~~ Furthermore, we
1134 show that S is strongly reduced in low vs. high elevation zones, suggesting that the conversion of
1135 recent OM inputs to stable compounds ~~and eventually SOM~~ is in fact lower in more flooded
1136 zones of tidal wetlands. ~~Accelerated~~ As the stabilization of labile OM inputs is a major driver of
1137 SOM formation (Cotrufo et al., 2013, 2015; Haddix et al., 2016), one important implication of
1138 this finding is that accelerated SLR consequently seems to yield yields the potential to decrease
1139 ~~SOM formation and with that C~~ the carbon-sequestration potential of tidal wetlands. This finding
1140 and its implication may occur counterintuitive with respect to the often sharp redox gradients
1141 along tidal wetland zonations and with flooding ~~The~~ ([Davy et al., 2011](#); [Kirwan et al., 2013](#);
1142 [Langley et al., 2013](#)), ~~and the~~ mechanism by which S is decreased in the more flooded zones of
1143 the present study is unknown. Because we did not observe consistent salinity effects on S and k
1144 in our data ([Figs. S1, S3](#)), we do not suppose that regular exposure of litter to salt water explains
1145 the unexpected finding. ~~Instead,~~ Likewise, soil temperature was not consistently affected by
1146 relative elevation across sites (p > 0.3; paired t-test based on data shown in Fig. S3). Instead, we
1147 argue that more favorable soil moisture conditions in low vs. high elevated zones could have
1148 decreased OM stabilization if higher flooding frequencies did not induce redox conditions low
1149 enough to suppress microbial activity in the top soil. In support of this, flooding-frequency
1150 induced changes in moisture conditions have been reported as primary driver of surface litter
1151 break-down, leading to more than four-fold increased litter mass loss in low vs. high marsh
1152 zones of a New Jersey salt marsh ([Halupa & Howes, 1995](#)).

1153 Additionally, greater nutrient availability and less nutrient-limited microbial communities
1154 in more frequently flooded zones could have contributed to this effect ([Deegan et al., 2012](#);

1155 Kirwan et al., 2013). Strong effects of both high quality marine-derived OM and nutrient
1156 amendments on microbial structure and activity have been reported (Deegan et al., 2012;
1157 ~~Keuskamp et al., 2015a~~; Kearns et al., 2016; [Keuskamp et al., 2015b](#); Mueller et al., 2017),
1158 suggesting that regular marine OM and nutrient inputs in more frequently flooded zones can
1159 positively affect decomposition. [\(see further discussed below in 4.3\)](#).

1160 4.3 Nutrient enrichment reduces stabilization – insights from the TIDE project

1161 In addition to our global survey of early-stage decomposition processes in tidal wetlands
1162 ~~worldwide~~, we included the long-term ecological research site of the TIDE project in
1163 Massachusetts, US to experimentally assess both the effects of coastal eutrophication and the
1164 relevance of nutrient delivery through floodwater for OM decomposition in tidal wetlands.
1165 Important for our argument that decomposition may be favored by higher nutrient availability in
1166 low elevated, more frequently flooded zones, we observed a strong reduction (>70%) of S by
1167 nutrient enrichment in the high marsh, whereas S in the low marsh likewise was low as in the
1168 fertilized high marsh and not further reduced by fertilization (Fig. [43d](#)). Johnson et al. (2016)
1169 demonstrate that nutrient enriched high-marsh plots of the TIDE project receive 19 ± 2 g N m⁻²
1170 yr⁻¹, approximately 10-times the N load of reference high-marsh plots (2 ± 1 g N m⁻² yr⁻¹;
1171 mean \pm SE), thus explaining the strong treatment effect observed in the high marsh. In accordance
1172 with low stabilization in the reference low marsh, which is equally low as the nutrient enriched
1173 high marsh, reference plots of the low marsh receive 16 ± 4 g N m⁻² yr⁻¹, the same high N load as
1174 the enriched high-marsh plots. Surprisingly, however, N loads of 171 ± 19 g N m⁻² yr⁻¹ in the
1175 enriched low-marsh plots do not result in additional reduction of S compared to the reference low
1176 marsh (Fig. [43d](#)). These findings suggest that microbial communities of the high marsh are N
1177 limited, and that N additions to a certain level can stimulate early OM decomposition and thus

1178 reduce stabilization. The missing effect of N loads exceeding 16 g m⁻² yr⁻¹ on stabilization in the
1179 low marsh indicates that microbial communities are less or not N limited due to the naturally
1180 greater nutrient availability. The findings of the TIDE project therefore support our concept that
1181 higher nutrient availability and less nutrient-limited microbial communities in more frequently
1182 flooded zones could have contributed to the observed reduction of OM stabilization in low vs.
1183 high elevated zones of tidal wetlands in our global assessment.

1184 Although our conclusions on effects of nutrient enrichment on OM decomposition are
1185 based on the findings of a single field experiment only, our study adds to a growing number of
1186 reports illustrating the impact of coastal eutrophication on tidal wetland C cycling (~~Morris &~~
1187 ~~Bradley, 1999~~; Deegan et al., 2012; ~~Keuskamp et al., 2015a~~; Kirwan ~~&~~ Megonigal, 2013;
1188 ~~Keuskamp et al., 2015b~~ Morris and Bradley, 1999). At the same time, however, we highlight the
1189 need to improve our understanding of coastal eutrophication in interaction with other global
1190 changes, particularly accelerated SLR and concomitant changes in flooding frequency, on the
1191 cycling of both labile and refractory C pools in order to predict future stability of tidal wetlands.

1192 4.4 ~~Methodological~~ The Tea Bag Index – methodological considerations

1193 ~~The~~ Interpretation of results obtained from standardized approaches like the present needs to be
1194 made cautiously because OM quality ~~of OM~~ (i.e. its chemical composition) is a key parameter
1195 affecting its decomposition. -As the quality of the TBI materials differ from that of wetland plant
1196 litters, and likely even more from the quality of the imported allochthonous OM (Khan et al.,
1197 2011), we did not expect to capture ~~precise and absolute values for wetland litter~~ actual rates of
1198 early-stage OM break-down in this study. Instead, we used the ~~Tea Bag Index~~ TBI to
1199 characterize the decomposition environment by obtaining a measure for the potential to
1200 decompose and stabilize the deployed standardized material. Standardized approaches like this,

1201 or also the cotton-strip assay (e.g. Latter and Walton, 1988), are useful to separate the effects of
1202 environmental factors other than OM quality on decomposition processes and to assess their
1203 relative importance. Otherwise, complex interaction effects of the abiotic environment and OM
1204 quality make it difficult to predict the relevance of certain environmental factors for
1205 decomposition processes, potentially masking the effects of important ~~global-change drivers~~
1206 ~~(reviewed in Prescott, 2010)~~global-change drivers (Prescott, 2010). At the same time, however,
1207 the global-change factors considered in the present study are likely to induce changes in the
1208 quality of the OM accumulating in tidal wetlands, for instance through shifts in plant-species
1209 composition and plant-tissue quality, that can potentially counterbalance or amplify the effects on
1210 decomposition processes suggested here. Future research therefore needs to address OM quality
1211 feedbacks on decomposition processes in tidal wetlands in order to gain a more complete
1212 understanding of global-change effects on tidal-wetland stability and carbon-sequestration
1213 capacity. Stabilization is thought to be a key parameter for capturing changes in decomposition
1214 with consequence for C sequestration. Indeed, Keuskamp et al. (2013) demonstrate that S, as
1215 determined by the TBI, is significantly related with the C sequestration potential of an ecosystem
1216 as defined by FAO (2000). In the present study, however, a large percentage of observations
1217 showed relatively low values for S, although tidal wetlands are known to act as particularly
1218 effective C sinks (Meleod et al., 2011). Based on the S values obtained from initial calculations
1219 using the ~~hydrolysable~~hydrolyzable fractions suggested by Keuskamp et al. (2013), a large
1220 number of observations ~~in fact~~ yielded a negative S (~~data not shown~~Table S3). S becomes
1221 negative when the mass loss from green tea is greater than the predicated maximum loss based on
1222 its ~~hydrolysable~~hydrolyzable fraction. At least two processes could have caused ~~or contributed to~~
1223 this result: First, ~~we demonstrate that S is indeed reduced in low vs. high elevated zones across~~
1224 our ~~study sites, indicating~~data indicate that redox conditions in the top soil of tidal wetlands are at

1225 ~~least often~~ not low enough to hamper decomposition ~~processes of the hydrolyzable fraction of the~~
1226 ~~TBI materials~~. As a consequence, ~~the relatively~~ high top-soil moisture of tidal wetlands ~~could~~
1227 provide favorable conditions for decomposition, following typical moisture-decomposition
1228 relationships as demonstrated for terrestrial ecosystems (e.g. Curiel Yuste et al., 2007), ~~and S~~
1229 ~~should at least not expected to be high in the top soil environment of tidal wetlands.~~ Potentially,
1230 moisture ~~conditions~~ and nutrient supply ~~are even high enough to~~ allow for considerable break-
1231 down of non-~~hydrolysable~~~~hydrolyzable~~ compounds within three months of deployment, such as
1232 lignin (Berg ~~&and~~ McClaugherty, 2003; ~~Knorr~~2014; ~~Duboc~~ et al., 2005~~2014~~; Feng et al., 2010;
1233 ~~Duboe~~~~Knorr~~ et al., 2014~~2005~~). Second, different protocols ~~and methods~~ to determine
1234 ~~hydrolysable and non-hydrolysable fractions~~~~the hydrolyzable fraction~~ of plant materials exist and
1235 lead to variable results. The ~~hydrolysable~~~~hydrolyzable~~ fraction ~~of plant materials~~ can
1236 consequently be over- or underestimated depending on ~~method~~, protocol, and type of sample
1237 material. The use of the slightly higher ~~hydrolysable~~~~hydrolyzable~~ fractions we determined for
1238 calculations of the TBI parameters effectively eliminated negative S values. In that regard, using
1239 the values obtained from the alternative protocol given in Ryan et al. (1990) seemed more
1240 reasonable in our study. ~~Although~~~~However, it needs to be stressed here that~~ direction and size of
1241 reported effects on S and k in the present study are almost independent ~~from~~~~of~~ the
1242 ~~hydrolysable~~~~hydrolyzable~~ fraction used for calculations, ~~further research is required to improve~~
1243 ~~our estimates of the hydrolysable fractions in TBI materials.~~

1244 Future research will have to test the applicability of the TBI approach in different
1245 ecosystems and test the validity of its assumptions (i.e. S is equal for both types of material used,
1246 and mass loss of non-hydrolyzable material is negligible over 3 months of deployment). The
1247 results of our regional scale assessment along the North American Atlantic coast transect are in
1248 tight agreement with previously reported results on cellulose break-down and soil microbial

1249 activity along this well studied transect (Kirwan et al., 2014; Mozdzer et al., 2014). We can
1250 thereby demonstrate the usefulness of the TBI approach to assess early-stage decomposition in
1251 tidal-wetland soils.

1252 4.5 Implications

1253 ~~While awareness about potential global warming impacts on OM preservation and their resulting~~
1254 ~~threat to future tidal wetland stability has been raised (Kirwan & Mudd, 2012), concepts on the~~
1255 ~~vulnerability of tidal wetlands to accelerated SLR mainly focus on plant productivity responses~~
1256 ~~and their biophysical feedbacks (Kirwan et al., 2016). Potentially negative effects of accelerated~~
1257 ~~SLR on OM preservation were thus far overlooked, probably because stimulation of~~
1258 ~~decomposition processes through increasing flooding is counterintuitive (Mueller et al., 2016).~~
1259 ~~Here, we provide evidence that accelerated SLR is unlikely to slow down the decomposition rate~~
1260 ~~of fresh OM inputs and additionally may strongly decrease OM stabilization and thus potentially~~
1261 ~~the fraction of net primary production and other OM inputs to stable SOM.~~

1262 This study addresses the influence of temperature, relative sea level, and coastal eutrophication
1263 on the initial transformation of biomass to SOM, and it does not encompass their effects on the
1264 existing SOM pool. However, aspects of S and k are key components of many tidal wetland
1265 resiliency models (Schile et al., 2014; Swanson et al., 2014) that have highlighted the critical role
1266 of the organic contribution to marsh elevation gain. ~~Thus, combined~~ Although actual rates of S
1267 and k cannot be inferred from this study using a standardized approach, our data identify strong
1268 negative effects of temperature, relative sea level, and coastal eutrophication on ~~OM~~ the
1269 stabilization ~~may~~ of fresh organic inputs to tidal-wetland soils. We argue that these unanticipated
1270 combined effects yield the potential to strongly ~~reduce OM accumulation rates and increase~~
1271 ~~wetland~~ accelerate carbon turnover in tidal wetlands, thus increasing their vulnerability to

1272 accelerated SLR, and we highlight the need for experimental studies assessing the extent to which
1273 the here identified effects translate into native OM dynamics.

1274 ~~Our findings imply that particularly the vulnerability of organic systems might increase with~~
1275 ~~global change because in these systems soil volume is almost exclusively generated by the~~
1276 ~~preservation of OM. At the same time, however, mineral dominated systems, such as temperate~~
1277 ~~European salt marshes, experience large amounts of easily decomposable allochthonous OM~~
1278 ~~input that relies on substantial stabilization in order to become sequestered (Middelburg et al.,~~
1279 ~~1997; Allen, 2000; Khan et al., 2015). Thus, future rates of C sequestration could be substantially~~
1280 ~~reduced in mineral dominated tidal wetland systems.~~

1281

1282

1283

1284

1285

1286

1287

1288

1289

1290

1291

1292

1293 **Acknowledgements**

1294 We thank Svenja Reents, Melike Yildiz, Anja Schrader, Detlef Böhm, Cailene Gunn, [Marcella](#)
1295 [Roner](#), Johan Krol, Marin van Regteren, Jacek Mazur, Ana Genua, Lluís Jornet, David Mateu,
1296 Sarah King, Shayne Levoy, and Lyntana Brougham for help with field and lab work. ~~This~~[The](#)
1297 project was [partly](#) funded by the Bauer-Hollmann Stiftung and the Rudolf und Helene Glaser
1298 Stiftung in the framework of the INTERFACE project. [This is contribution 25 from the](#)
1299 [Smithsonian's MarineGEO Network](#). The authors declare no conflict of interest.

1300 **Author contributions**

1301 PM, SN, KJ, and LMS-B designed the overall study. PM analyzed and interpreted the data. PM
1302 wrote the initial version of the manuscript with regular comments and editing provided by LMS-
1303 B, TJM, and SN. PM, LMS-B, TJM, GLC, TD, YK, AVdG, PE, CS, AD'A, CI, ML, UN, BJJ,
1304 AHB, SAY, DIM, ZY, and JW designed and conducted the field studies in the respective sites
1305 and commented on an earlier version of the manuscript.

1306

1307

1308

1309

1310

1311

1312

1313

1314

1315

1316

1317

1318

1319

1320

1321

1322 **Figure captions**

1323 Figure 1 Overview map of study sites/regions. Notes: See Table 1 for region and site details.

1324 Figure 2 (a) Decomposition rate (k site means; n = 30) and (b) stabilization factor (S site means; n
1325 = 30) versus mean atmospheric air temperature during deployment period. Shown are results of
1326 linear regression line illustrates significant relationship between temperature and stabilization
1327 (Table 2); regression analyses across and within elevation zones and organic model with lowest
1328 standard error of estimate (SEE) and mineral soils. highest R² is shown: $y = -0.27\ln(x) + 0.99$; R^2
1329 = 0.239; SEE = 0.131; excluding the Mediterranean sites (21.9-23.6°C; n = 4) from the regression
1330 yields: $y = -0.344\ln(x) + 1.233$; $R^2 = 0.510$; SEE = 0.101

1331 Figure 3 (a + e) Decomposition rate (k) and (b + d) stabilization factor (S) in high (orange) and
1332 low (blue) elevated zones of tidal marsh and mangrove sites (compare Table 1). High and low
1333 elevated zones are characterized by distinct plant species assemblages or by different stature of
1334 mangroves along the flooding gradient within each site. Shown are means ± SE for all sites (a +
1335 b) and for mineral, organic, marsh, and mangrove systems separately (c + d). P values refer to
1336 results of paired t tests (ns, $P > 0.05$; * $P \leq 0.05$; ** $P \leq 0.01$).

1337 Figure 4 Effects of marsh elevation (zone) and n = 21; compare Tables 1, S1). (c) Decomposition
1338 rate and (d) stabilization in nutrient enrichment on both decomposition rate (k) and stabilization
1339 factor (S) in long term enriched (filled bars) and versus reference areas (open bars) in the high
1340 marsh (Spartina patens zone) and low marsh (Spartina alterniflora zone) of the TIDE project site
1341 at the Plum Island Sound Estuary, Massachusetts, US. Shown are means ± SE and results of
1342 paired t-tests (panels a + b) and two-way ANOVAs and plus Tukey's HSD test for pairwise
1343 comparisons (Tukey's HSD test). panels c + d): ns = not significant; * = $p \leq 0.05$; ** = $p \leq 0.01$

1344

1345

1346

1347

1348

1349

1350

1351

1352

1353

1354 **References**

1355 [Allen, J. R. L.: Morphodynamics of Holocene salt marshes: A review sketch from the Atlantic and Southern North Sea coasts of Europe, *Quat. Sci. Rev.*, 19\(12\), 1155–1231, doi:10.1016/S0277-3791\(99\)00034-7, 2000.](#)

1356

1357 Baldwin, A. H., Jensen, K. and Schönfeldt, M.: Warming increases plant biomass and reduces diversity across continents, latitudes, and species migration scenarios in experimental wetland communities, *Glob. Chang. Biol.*, 20(3), 835–850, doi:10.1111/gcb.12378, 2014.

1358

1359 Benito, X., Trobajo, R. and Ibáñez, C.: Modelling habitat distribution of Mediterranean coastal wetlands: The Ebro delta as case study, *Wetlands*, 34(4), 775–785, doi:10.1007/s13157-014-0541-2, 2014.

1360

1361 Berg, B. and McLaugherty, C.: *Plant litter: Decomposition, humus formation, carbon sequestration*, 2003, 3rd ed., Springer-Verlag, Berlin/Heidelberg., 2014.

1362

1363 [Bernal, B., Magonigal, J. P. and Mozdzer, T. Berg, B., Berg, M. P., Bottner, P., Box, E., Breymer, A., de Anta, R. C., Couteaux, M., Escudero, A., Gallardo, A., Kratz, W., Madeira, M., Mäkönen, E., McLaugherty, C., Meentemeyer, V., Muñoz, F., Piussi, P., Remacle, J. and de Santo, A. V.: Litter mass loss rates in pine forests of Europe and Eastern United States: some relationships with climate and litter quality, *Biogeochemistry*, 20\(3\), 127–159, doi:10.1007/BF00000785, 1993.](#)

1364

1365 [J., An Invasive Wetland Grass Primes Deep Soil Carbon Pools, *Glob. Chang. Biol.*, 2104–2116, doi:10.1111/gcb.13539, 2017.](#)

1366

1367 [Boot, R. G. A. and Mensink, M.: Size and Morphology of Root Systems of Perennial Grasses From Contrasting Habitats As Affected By Nitrogen Supply, *Plant Soil*, 129\(2\), 291–299 \[online\] Available from: http://apps.webofknowledge.com/full_record.do?product=UA&search_mode=GeneralSearch&qid=7&SID=2BS7zGFM4K9BDD](#)

1368

1369 [eZbHg&page=1&doc=4, 1990.](#)

1370

1371 [Bradford, M. A., Tordoff, G. M., Eggers, T., Jones, T. H. and Newington, J. E.: Microbiota, fauna, and mesh size interactions in litter decomposition, *Oikos*, 99\(2\), 317–323, doi:10.1034/j.1600-0706.2002.990212.x, 2002.](#)

1372

1373 [Breiman, L., Friedman, J. H., Olshen, R. A. and Stone, C. J.: *Classification and Regression Trees*, Wadsworth International Group, Belmont, CA, USA., 1984.](#)

1374

1375

1376 Brinkmann, K., Blaschke, L. and Polle, A.: Comparison of different methods for lignin determination as a basis for calibration of near-infrared reflectance spectroscopy and implications of lignoproteins., *J. Chem. Ecol.*, 28(12), 2483–501 [online] Available from: <http://www.ncbi.nlm.nih.gov/pubmed/12564795>, 2002.

1377

1378 Castellano, M. J., Mueller, K. E., Olk, D. C., Sawyer, J. E. and Six, J.: Integrating plant litter quality, soil organic matter stabilization, and the carbon saturation concept, *Glob. Chang. Biol.*, 21(9), 3200–3209, doi:10.1111/gcb.12982, 2015.

1379

1380 Chang, E. R., Veeneklaas, R. M., Bakker, J. P., Daniels, P. and Esselink, P.: What factors determined restoration success of a salt marsh ten years after de-embankment?, *Appl. Veg. Sci.*, 19(1), 66–77, doi:10.1111/avsc.12195, 2016.

1381

1382 Charles, H. and Dukes, J. S.: Effects of warming and altered precipitation on plant and nutrient dynamics of a New England salt marsh, *Ecol. Appl.*, 19(7), 1758–1773, doi:10.1890/08-0172.1, 2009.

1383

1384 Chmura, G., Chase, P. and Bercovitch, J.: Climatic Controls of the Middle Marsh Zone in the Bay of Fundy, *Estuaries*, 20(4), 689–699, 1997.

1385

1386 Chmura, G. L., Anisfeld, S. C., Cahoon, D. R. and Lynch, J. C.: Global carbon sequestration in tidal, saline wetland soils, *Global Biogeochem. Cycles*, 17(4), 22-1-22–11, doi:10.1029/2002GB001917, 2003.

1387

1388 Cotrufo, M. F., Wallenstein, M. D., Boot, C. M., Deneff, K. and Paul, E.: The Microbial Efficiency-Matrix Stabilization (MEMS) framework integrates plant litter decomposition with soil organic matter stabilization: Do labile plant inputs form stable soil organic matter?, *Glob. Chang. Biol.*, 19(4), 988–995, doi:10.1111/gcb.12113, 2013.

1389

1390 [GraineCotrufo, M. F., Soong, J. L., Horton, A. J., Campbell, E. E., Haddix, M. L., M., Fierer, N., Wall, D. H. and Parton, W. J.: Widespread coupling between the rate and temperature sensitivity of soil organic matter decay via biochemical and physical pathways of litter mass loss, *Nat. Geosci.*, 3\(12\), 854–857, 776–779, doi:10.1038/ngeo2520, 2015.](#)

1391

1392 [Craft, C., Clough, J., Ehman, J., Jove, S., Park, R., Pennings, S., Guo, H. and Machmuller, M.: Forecasting the effects of accelerated sea-level rise on tidal marsh ecosystem services, *Front. Ecol. Environ.*, 7\(2\), 73–78, doi:10.1890/070219.2009.](#)

1393

1394 [Crosby, S. C., Sax, D. F., Palmer, M. E., Booth, H. S., Deegan, L. A., Bertness, M. D. and Leslie, H. M.: Salt marsh persistence is threatened by predicted sea-level rise, *Estuar. Coast. Shelf Sci.*, ngeo1009, 2010181, 93–99, doi:10.1016/j.ecss.2016.08.018, 2016.](#)

1395

1396 Curiel Yuste, J., Baldocchi, D. D., Gershenson, A., Goldstein, A., Misson, L. and Wong, S.: Microbial soil respiration and its dependency on carbon inputs, soil temperature and moisture, *Glob. Chang. Biol.*, 13(9), 2018–2035, doi:10.1111/j.1365-2486.2007.01415.x, 2007.

1397

1398 Davidson, E. A. and Janssens, I. A.: Temperature sensitivity of soil carbon decomposition and feedbacks to climate change., *Nature*, 440(7081), 165–73, doi:10.1038/nature04514, 2006.

1399

1400 Davy, A. J., Brown, M. J. H., Mossman, H. L. and Grant, A.: Colonization of a newly developing salt marsh: Disentangling independent effects of elevation and redox potential on halophytes, *J. Ecol.*, 99(6), 1350–1357, doi:10.1111/j.1365-2745.2011.01870.x, 2011.

1401

1402 Deegan, L. A., Johnson, D. S., Warren, R. S., Peterson, B. J., Fleeger, J. W., Fagherazzi, S. and Wollheim, W. M.: Coastal eutrophication as a driver of salt marsh loss, *Nature*, 490, 338–392, doi:10.1038/nature11533, 2012.

1403

1404 Dijkema, K. S., Kers, A. S. and Duin, W. E. Van: Salt marshes: applied long-term monitoring salt marshes, *Wadden Sea Ecosyst.*, (26), 35–40, 2010.

1405

1406 [Djukic, I., Kepfer-Rojas, S., Schmidt, I. K., Larsen, K. S., Beier, C., Berg, B., Verheyen, K., Caliman, A., Paquette, A., Gutiérrez-Girón, A., Humber, A., Valdecantos, A., Petraglia, A., Alexander, H., Augustaitis, A., Saillard, A., Fernández, A. C. R., Sousa, A. I., Lillebø, A. I., da Rocha Gripp, A., Francez, A.-J., Fischer, A., Bohner, A., Malyshev, A., Andrić, A., Smith, A., Stanisci, A., Seres, A., Schmidt,](#)

1407

1408

1409

1410

1411

1412

1413

1414

1415

1416 [A., Avila, A., Probst, A., Ouin, A., Khuroo, A. A., Verstraeten, A., Palabral-Aguilera, A. N., Stefanski, A., Gaxiola, A., Muys, B.,](#)
1417 [Bosman, B., Ahrends, B., Parker, B., Sattler, B., Yang, B., Juráni, B., Erschbamer, B., Ortiz, C. E. R., Christiansen, C. T., Carol Adair,](#)
1418 [E., Meredieu, C., Mony, C., Nock, C. A., Chen, C.-L., Wang, C.-P., Baum, C., Rixen, C., Delire, C., Piscart, C., Andrews, C., Rebmann,](#)
1419 [C., Branquinho, C., Polyanskaya, D., Delgado, D. F., Wundram, D., Radeideh, D., Ordóñez-Regil, E., Crawford, E., Preda, E.,](#)
1420 [Tropina, E., Groner, E., Lucot, E., Hornung, E., Gacia, E., Lévesque, E., Benedito, E., Davydov, E. A., Ampoorter, E., Bolzan, F. P.,](#)
1421 [Varela, F., Kristöfel, F., Maestre, F. T., Maunoury-Danger, F., Hofhansl, F., Kitz, F., Sutter, F., Cuesta, F., de Almeida Lobo, F., de](#)
1422 [Souza, F. L., Berninger, F., Zehetner, F., Wohlfahrt, G., Vourlitis, G., Carreño-Rocabado, G., Arena, G., Pinha, G. D., González, G.,](#)
1423 [Canut, G., Lee, H., Verbeeck, H., Auge, H., et al.: Early stage litter decomposition across biomes, *Sci. Total Environ.*, 628–629,](#)
1424 [1369–1394, doi:10.1016/j.scitotenv.2018.01.012, 2018.](#)
1425 Duboc, O., Dignac, M. F., Djukic, I., Zehetner, F., Gerzabek, M. H. and Rumpel, C.: Lignin decomposition along an Alpine elevation
1426 gradient in relation to physicochemical and soil microbial parameters, *Glob. Chang. Biol.*, 20(7), 2272–2285,
1427 doi:10.1111/gcb.12497, 2014.
1428 Erland, B. and Soderstrom, B.: The significance of hyphal diameter in calculation of fungal biovolume, , 33(1), 11–14, 1979.
1429 Feng, X., Simpson, A. J., Schlesinger, W. H. and Simpson, M. J.: Altered microbial community structure and organic matter
1430 composition under elevated CO₂ and N fertilization in the duke forest, *Glob. Chang. Biol.*, 16(7), 2104–2116, doi:10.1111/j.1365-
1431 2486.2009.02080.x, 2010.
1432 Flemming, B. W. and Davis, R. A.: Holocene evolution, morphodynamics and sedimentology of the Spiekeroog barrier island
1433 system (southern North Sea), *Senckenbergia maritima*, 24((1/6)), 117–155, 1994.
1434 Gedan, K. B. and Bertness, M. D.: Experimental warming causes rapid loss of plant diversity in New England salt marshes, *Ecol.*
1435 *Let.*, 12(8), 842–848, doi:10.1111/j.1461-0248.2009.01337.x, 2009.
1436 Haddix, M. L., Paul, E. A. and Cotrufo, M. F.: Dual, differential isotope labeling shows the preferential movement of labile plant
1437 constituents into mineral-bonded soil organic matter, *Glob. Chang. Biol.*, 22(6), 2301–2312, doi:10.1111/gcb.13237, 2016.
1438 Halupa, P. J. and Howes, B. L.: Effects of tidally mediated litter moisture content on decomposition of *Spartina alterniflora* and *S.*
1439 *patens*, *Mar. Biol.*, 123(2), 379–391, doi:10.1007/BF00353629, 1995.
1440 ~~Hines, J., Reyes, M., Mozdzer, T. J. and Gessner, M. O.: Genotypic trait variation modifies effects of climate warming and nitrogen~~
1441 ~~deposition on litter mass loss and microbial respiration, *Glob. Chang. Biol.*, doi:10.1111/gcb.12704, 2014.~~
1442 Howison, R. A., Olf, H., Steever, R. and Smit, C.: Large herbivores change the direction of interactions within plant communities
1443 along a salt marsh stress gradient, *J. Veg. Sci.*, 26(6), 1159–1170, doi:10.1111/jvs.12317, 2015.
1444 Isacch, J., Costa, C., Rodriguez-Gallego, L., Conde, D., Escapa, M., Gagliardini, D. and Iribarne, O.: Distribution of saltmarsh plant
1445 communities associated with environmental factors along a latitudinal gradient on the south-west Atlantic coast, *Journal of*
1446 *Biogeography*, 33(5), 888–900, doi:10.1111/j.1365-2699.2006.01461.x, 2006.
1447 Janousek, C. N., Buffington, K. J., Guntenspergen, G. R., Thorne, K. M., Dugger, B. D. and Takekawa, J. Y.: Inundation, vegetation,
1448 and sediment effects on litter decomposition in Pacific Coast tidal marshes, *Ecosystems*, 2014–2015, doi:10.1007/s10021-017-
1449 0111-6, 2017.
1450 Johnson, D. S., Warren, R. S., Deegan, L. A. and Mozdzer, T. J.: Saltmarsh plant responses to eutrophication, *Ecol. Appl.*, 26(8),
1451 2647–2659, doi:10.1002/eap.1402, 2016.
1452 Kearns, P. J., Angell, J. H., Howard, E. M., Deegan, L. A., Stanley, R. H. R. and Bowen, J. L.: Nutrient enrichment induces dormancy
1453 and decreases diversity of active bacteria in salt marsh sediments, *Nat. Commun.*, 7, 1–9, doi:10.1038/ncomms12881, 2016.
1454 Keuskamp, J. A., Dingemans, B. J. J., Lehtinen, T., Sarneel, J. M. and Hefting, M. M.: Tea Bag Index: a novel approach to collect
1455 uniform decomposition data across ecosystems, *Methods Ecol. Evol.*, 4(11), 1070–1075, doi:10.1111/2041-210X.12097, 2013.
1456 Keuskamp, J. A., Hefting, M. M., Dingemans, B. J. J., Verhoeven, J. T. A. and Feller, I. C.: Effects of nutrient enrichment on
1457 mangrove leaf litter decomposition, *Sci. Total Environ.*, doi:10.1016/j.scitotenv.2014.11.092, 2015a.
1458 Keuskamp, J. A., Feller, I. C., Laanbroek, H. J., Verhoeven, J. T. A. and Hefting, M. M.: Short- and long-term effects of nutrient
1459 enrichment on microbial exoenzyme activity in mangrove peat, *Soil Biol. Biochem.*, 81(NOVEMBER), 38–47,
1460 doi:10.1016/j.soilbio.2014.11.003, 2015b.
1461 Khan, N. S., Vane, C. H. and Horton, B. P.: Stable carbon isotope and C/N geochemistry of coastal wetland sediments as a sea-
1462 level indicator, *Handb. Sea-Level Res.*, 295–311, doi:10.1002/9781118452547.ch20, 2015.
1463 Kirwan, M. L. and Blum, L. K.: Enhanced decomposition offsets enhanced productivity and soil carbon accumulation in coastal
1464 wetlands responding to climate change, *Biogeosciences*, 8(4), 987–993, doi:10.5194/bg-8-987-2011, 2011.
1465 Kirwan, M. L. and Megonigal, J. P.: Tidal wetland stability in the face of human impacts and sea-level rise, *Nature*, 504(7478),
1466 53–60, doi:10.1038/nature12856, 2013.
1467 ~~Kirwan, M. L. and Mudd, S. M.: Response of salt marsh carbon accumulation to climate change., *Nature*, 489(7417), 550–3,~~
1468 ~~doi:10.1038/nature11440, 2012.~~
1469 Kirwan, M. L., Guntenspergen, G. R. and Morris, J. T.: Latitudinal trends in *Spartina alterniflora* productivity and the response of
1470 coastal marshes to global change, *Glob. Chang. Biol.*, 15(8), 1982–1989, doi:10.1111/j.1365-2486.2008.01834.x, 2009.
1471 Kirwan, M. L., Langley, J. A., Guntenspergen, G. R. and Megonigal, J. P.: The impact of sea-level rise on organic matter decay
1472 rates in Chesapeake Bay brackish tidal marshes, *Biogeosciences*, 10(3), 1869–1876, doi:10.5194/bg-10-1869-2013, 2013.
1473 Kirwan, M. L., Guntenspergen, G. R. and Langley, J. A.: Temperature sensitivity of organic-matter decay in tidal marshes,
1474 *Biogeosciences*, 11, 4801–4808, doi:10.5194/bg-11-4801-2014, 2014.

1475 [Kirwan, M., Temmerman, S., Skeechean, E. E., Guntenspergen, G. R. and Faghe, S.: Overestimation of marsh vulnerability to sea](#)
1476 [level rise, *Nat. Clim. Chang.*, 6, 253–260, doi:10.1038/nclimate2909, 2016.](#)

1477 Knorr, M., Frey, S. D. and Curtis, P. S.: Nitrogen additions and litter decomposition: A meta analysis, *Ecology*, 86(12), 3252–3257,
1478 doi:10.1890/05-0150, 2005.

1479 [Kristensen, E., Ahmed, S. I. and Devol, A. H.: Aerobic and anaerobic decomposition of organic matter in marine sediment: Which](#)
1480 [is fastest?, *Limnol. Oceanogr.*, 40\(8\), 1430–1437, doi:10.4319/lo.1995.40.8.1430, 1995.](#)

1481 Latter, P.M. and Walton, D.W.H.: The cotton strip assay for cellulose decomposition studies in soil: history of the assay and
1482 development, in: Cotton strip assay: an index of decomposition in soils, edited by: Harrison, A.F., Latter, P.M. and Walton,
1483 D.W.H., Grange-over-Sands, UK, 1988.

1484 Langley, J. A. and Megonigal, J. P.: Ecosystem response to elevated CO₂ levels limited by nitrogen-induced plant species shift, *Nature*,
1485 466(7302), 96–99, doi:10.1038/nature09176, 2010.

1486 Langley, J. A., Mozdzer, T. J., Shepard, K. A., Hagerty, S. B. and Megonigal, J. P.: Tidal marsh plant responses to elevated CO₂,
1487 nitrogen fertilization, and sea level rise, *Glob. Chang. Biol.*, 19(5), 1495–503, doi:10.1111/gcb.12147, 2013.

1488 Lovelock, C. E., Feller, I. C., McKee, K. L. and Thompson, R.: Variation in mangrove forest structure and sediment characteristics
1489 in Bocas del Toro, Panama, *Caribb. J. Sci.*, 41(3), 456–464, doi:ISSN 0008-6452, 2005.

1490 [Macreadie, P. I., Nielsen, D. A., Kelleway, J. J., Atwood, T. B., Seymour, J. R., Petrou, K., Connolly, R. M., Thomson, A. C.,](#)
1491 [Trevathan-Tackett, S. M. and Ralph, P. J.: Can we manage coastal ecosystems to sequester more blue carbon?, *Front. Ecol.*](#)
1492 [Environ.](#), in press, doi:10.1002/fee.1484, 2017.

1493 McKee, K. L., Cahoon, D. R. and Feller, I. C.: Caribbean mangroves adjust to rising sea level through biotic controls on change in
1494 soil elevation, *Glob. Ecol. Biogeogr.*, 16(5), 545–556, doi:10.1111/j.1466-8238.2007.00317.x, 2007.

1495 Mcleod, E., Chmura, G. L., Bouillon, S., Salm, R., Björk, M., Duarte, C. M., Lovelock, C. E., Schlesinger, W. H. and Silliman, B. R.: A
1496 blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO₂,
1497 *Front. Ecol. Environ.*, 9(10), 552–560, doi:10.1890/l, 2011.

1498 [McTiernan, K. B., Coûteaux, M. M., Middelburg, J. J., Nieuwenhuize, J., Lubberts, R. K. and Plassche, O. Van De: Organic Carbon](#)
1499 [Isotope Systematics of Coastal Marshes, *Estuar. Berg, B., Berg, M. P., De Anta, R. C., Gallardo, A., Kratz, W., Piuksi, P., Remacle, J.*](#)
1500 [and De Santo, A. V.: Changes in chemical composition of Pinus sylvestris needle litter during decomposition along a European](#)
1501 [coniferous forest climatic transect, *Soil Biol. Biochem.*, doi:10.1016/S0038-0717\(03\)00107-X, 2003.](#)

1502 [Coast. Shelf Sci.](#), 45, 681–687, 1997.

1503 Morris, J. and Bradley, P.: Effects of nutrient loading on the carbon balance of coastal wetland sediments, *Limnol. Oceanogr.*,
1504 44(3)(3), 699–702, doi:10.4319/lo.1999.44.3.0699, 1999.

1505 Morris, J. T., Sundberg, K. and Hopkinson, C. S.: Salt marsh primary production and its responses to relative sea level and
1506 nutrients, *Oceanography*, 26(3), 78–84, 2013.

1507 [Mozdzer, T. J., McGlathery, K. J., Mills, A. L. and Zieman, J. C.: Latitudinal variation in the availability and use of dissolved organic](#)
1508 [nitrogen in Atlantic coast salt marshes, *Ecology*, 95\(12\), 3293–3303, 2014.](#)

1509 Mueller, P., Jensen, K. and Megonigal, J. P.: Plants mediate soil organic matter decomposition in response to sea level rise, *Glob.*
1510 *Chang. Biol.*, 22(1), 404–414, doi:10.1111/gcb.13082, 2016.

1511 Mueller, P., Granse, D., Nolte, S., Do, H. T., Weingartner, M., Hoth, S. and Jensen, K.: Top-down control of carbon sequestration:
1512 grazing affects microbial structure and function in salt marsh soils, *Ecol. Appl.*, in press 27(5), 1435–1450, doi:10.1002/eap.1534,
1513 2017.

1514 Neff, K. P., Rusello, K. and Baldwin, A. H.: Rapid seed bank development in restored tidal freshwater wetlands, *Restor. Ecol.*,
1515 17(4), 539–548, doi:10.1111/j.1526-100X.2008.00415.x, 2009.

1516 Neumeier, U. and Cheng, C.: Hydrodynamics and sediment dynamics in an ice-covered tidal flat, *Coast. Sediments* 2015, 1–14,
1517 doi:10.1142/9789814689977, 2015.

1518 Nolte, S., Müller, F., Schuerch, M., Wanner, A., Esselink, P., Bakker, J. P. and Jensen, K.: Does livestock grazing affect sediment
1519 deposition and accretion rates in salt marshes?, *Estuar. Coast. Shelf Sci.*, 135 (July 2016), 296–305,
1520 doi:10.1016/j.ecss.2013.10.026, 2013.

1521 [Pendleton, L., Donato, D. C., Murray, B. C., Crooks, S., Jenkins, W. A., Sifleet, S., Craft, C., Fourqurean, J. W., Kauffman, J. B.,](#)
1522 [Marbà, N., Megonigal, P., Pidgeon, E., Herr, D., Gordon, D. and Baldera, A.: Estimating global “blue carbon” emissions from](#)
1523 [conversion and degradation of vegetated coastal ecosystems., *PLoS One*, 7\(9\), e43542, doi:10.1371/journal.pone.0043542,](#)
1524 [2012.](#)

1525 [Powers, J. S., Montgomery, R. A., Adair, E. C., Brearley, F. Q., Dewalt, S. J., Castanho, C. T., Chave, J., Deinert, E., Ganzhorn, J. U.,](#)
1526 [Gilbert, M. E., González-Iturbe, J. A., Bunyavejchewin, S., Grau, H. R., Harms, K. E., Hiremath, A., Iriarte-Vivar, S., Manzano, E., De](#)
1527 [Oliveira, A. A., Poorter, L., Ramanamanjato, J. B., Salk, C., Varela, A., Weiblen, G. D. and Lerdau, M. T.: Decomposition in tropical](#)
1528 [forests: A pan-tropical study of the effects of litter type, litter placement and mesofaunal exclusion across a precipitation](#)
1529 [gradient, *J. Ecol.*, doi:10.1111/j.1365-2745.2009.01515.x, 2009.](#)

1530 Prescott, C. E.: Litter decomposition: What controls it and how can we alter it to sequester more carbon in forest soils?,
1531 *Biogeochemistry*, 101(1), 133–149, doi:10.1007/s10533-010-9439-0, 2010.

1532 Roner, M., D’Alpaos, A., Ghinassi, M., Marani, M., Silvestri, S., Franceschinis, E. and Realdon, N.: Spatial variation of salt-marsh
1533 organic and inorganic deposition and organic carbon accumulation: Inferences from the Venice lagoon, Italy, *Adv. Water*
1534 *Resour.*, 93(November), 276–287, doi:10.1016/j.advwatres.2015.11.011, 2016.

1535 Rothwell, J. J., Futter, M. N. and Dise, N. B.: A classification and regression tree model of controls on dissolved inorganic nitrogen
1536 leaching from European forests, *Environ. Pollut.*, 156(2), 544–552, doi:10.1016/j.envpol.2008.01.007, 2008.
1537 Ryan, M. G., Melillo, J. M. and Ricca, A.: A comparison of methods for determining proximate carbon fractions of forest litter,
1538 *Can. J. For. Res. Can. Rech. For.*, 20(2), 166–171, doi:10.1139/x90-023, 1990.
1539 Schile, L. M., Callaway, J. C., Morris, J. T., Stralberg, D., Thomas Parker, V. and Kelly, M.: Modeling tidal marsh distribution with
1540 sea-level rise: Evaluating the role of vegetation, sediment, and upland habitat in marsh resiliency, *PLoS One*, 9(2), e88760,
1541 doi:10.1371/journal.pone.0088760, 2014.
1542 Soil Survey Staff: Keys to soil taxonomy, *Soil Conserv. Serv.*, 2014.
1543 Swanson, K. M., Drexler, J. Z., Schoellhamer, D. H., Thorne, K. M., Casazza, M. L., Overton, C. T., Callaway, J. C. and Takekawa, J.
1544 Y.: Wetland Accretion Rate Model of Ecosystem Resilience (WARMER) and its application to habitat sustainability for
1545 endangered species in the San Francisco Estuary, *Estuaries and Coasts*, 37(2), 476–492, doi:10.1007/s12237-013-9694-0, 2014.
1546 [Trofymow, J., Moore, T., Titus, B., Prescott, C., Morrison, I., Siltanen, M., Smith, S., Fyles, J., Wein, R., Camiré, C., Duschene, L.,](#)
1547 [Kozak, L., Kranabetter, M. and Visser, S.: Rates of litter decomposition over 6 years in Canadian forests: influence of litter quality](#)
1548 [and climate, *Can. J. For. Res.*, doi:10.1139/x01-117, 2002.](#)
1549 Vasey, M. C., Parker, V. T., Callaway, J. C., Herbert, E. R. and Schile, L. M.: Tidal wetland vegetation in the San Francisco Bay-Delta
1550 Estuary, *San Fr. Estuary Watershed Sci.*, 10(2), 1–16, 2012.
1551 ~~Wilson, R. M., Hopple, A. M., Tfaily, M. M., Sebestyen, S. D., Schadt, C. W., Pfeifer-Meister, L., Medvedeff, C., McFarlane, K.,~~
1552 ~~Kostka, J. E., Kolton, M., Kolka, R. K., Kluber, L. A., Keller, J. K., Guilderson, T. P., Griffiths, N. A., Chanton, J. P., Bridgham, S. D.~~
1553 ~~and Hanson, P. J.: Stability of peatland carbon to rising temperatures, *Nat. Commun.*, 7, #13723, doi:10.1038/ncomms13723,~~
1554 ~~2016.~~
1555 ~~Wolf, A. A., Drake, B. G., Erickson, J. E. and Megonigal, J. P.: An oxygen-mediated positive feedback between elevated carbon~~
1556 ~~dioxide and soil organic matter decomposition in a simulated anaerobic wetland, *Glob. Chang. Biol.*, 13(9), 2036–2044,~~
1557 ~~doi:10.1111/j.1365-2486.2007.01407.x, 2007.~~
1558 Yang, Z., Nolte, S. and Wu, J.: Tidal flooding diminishes the effects of livestock grazing on soil micro-food webs in a coastal
1559 saltmarsh, *Agric. Ecosyst. Environ.*, 236, 177–186, doi:10.1016/j.agee.2016.12.006, 2017.
1560 [Zhang, D. Q., Hui, D. F., Luo, Y. Q. and Zhou, G. Y.: Rates of litter decomposition in terrestrial ecosystems: global patterns and](#)
1561 [controlling factors, *J. Plant Ecol.*, 1\(2\), 85–93, doi:10.1093/jpe/rtn002, 2008.](#)
1562
1563
1564