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1 **Agroecosystem resilience in response to extreme winter flooding**

2

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15

16 **Highlights**

- 17 • Extreme winter flooding negatively altered soil physical, chemical and biological
18 indicators.
- 19 • Soil available P was reduced by 42% in the flooded areas after the flood event.
- 20 • Plant biomass was reduced by 0 or 19-34% in flooded areas.
- 21 • Total soil microbial biomass was increased by 60% after flooding.
- 22 • Grassland soils were more resilient than other crops.

23

24 **Abstract**

25 Evidence suggests that climate change is increasing the frequency of extreme weather events
26 (e.g. excessive rainfall, heat, wind). The winter of 2013-14 saw exceptional levels of rainfall
27 across the UK leading to extreme and prolonged flooding (up to 3 months with floodwater
28 depths up to 3 m) in several low-lying agricultural areas (e.g. Somerset Levels, Thames
29 Valley). The impact of extreme flooding and the speed of ecosystem recovery at the field-scale,
30 however, remain poorly understood. The main objectives of this study were therefore to: (1)
31 assess the effect of this extreme winter flooding event on a range of soil physical, chemical and
32 biological quality indicators at 15 flood-affected sites (arable and grassland), (2) determine if
33 these changes in soil health were reversible in the short term (< 1 year), and (3) to evaluate the
34 effectiveness of different mechanical interventions (sward-lifting, subsoiling, slot-seeding and
35 aerating) to accelerate the amelioration of the damage caused by winter flooding at 2 of the 15
36 sites. Once the floodwater had receded (April 2014), we found that several of the measured soil
37 quality indicators were negatively affected in the flooded areas in comparison with non-flooded
38 areas. This included a decrease in soil bulk density (by 19%), soil pH (by 0.4 units), and
39 available P (by up to 42%). Flooding increased soil microbial biomass (60%), induced a shift
40 in soil microbial community structure and reduced earthworm numbers. After 8 months of
41 recovery, only soil pH remained significantly reduced (by 0.3 units) in the flooded areas in
42 comparison to the unflooded areas. Flooding had a negative impact on the overlying vegetation
43 at the arable sites (biomass production was reduced by between 19 and 34%) but had no major
44 impact at the grassland sites in the long-term. In the flood amelioration experiment, the
45 subsoiled plots produced grass with a higher nutrient content (e.g. N - up to 35%, Ca - up to
46 19% and Mg - up to 58%). However, the four different interventions appeared to have little
47 positive impact on most of the soil quality indicators measured. In conclusion, extreme winter
48 flooding was found to induce short-term alterations in key soil quality indicators and to destroy

49 winter crops, although these effects did not persist in the longer term. Our results therefore
50 indicate that the temperate agroecosystems evaluated here were highly resilient to winter flood
51 stress and that recovery to a pre-flood state could be achieved within 1 year. Improved
52 management strategies are still needed to speed up the rate of recovery after flood events to
53 facilitate a faster return to agricultural production.

54

55 **Keywords:** Extreme weather; Nutrient cycling; PLFAs; Waterlogging.

56

57 **1. Introduction**

58 There is increasing evidence that short-term extreme weather events (e.g. excessive rainfall,
59 heat, wind) are becoming increasingly frequent globally (Donat et al., 2016), potentially leading
60 to negative effects (i.e. floods, droughts) and threatening long-term terrestrial ecosystem
61 functioning (Harris et al., 2018). These increases are more evident in North America and
62 Europe in comparison with other countries located in the Southern Hemisphere (Berghuijs et
63 al., 2017). For example, the winter of 2013-2014 saw exceptional levels of rainfall in the UK
64 leading to extreme and prolonged flooding in many low lying areas with agricultural land
65 remaining under water for up to 3 months (Slingo et al., 2014; Defra, 2014). Similar events
66 have occurred in other countries such as the USA in 2011, 2013 and 2014 (Mallakpour and
67 Villarini, 2015).

68 Perhaps the most obvious impact of prolonged flooding in agricultural fields is the
69 damage to crops (Malik et al., 2002). Soil becomes anaerobic when it is waterlogged, and this
70 has almost immediate effects on vegetation. Within 48 h, plants begin to suffer from O₂
71 deprivation, which causes a significant reduction in nutrient uptake rates, inhibiting plant
72 growth both above and belowground (Jackson, 2004). If waterlogged or anaerobic conditions
73 persist, hydrogen sulphide, acetic acid and butyric acid are produced as the soil redox potential

74 levels reduce. These compounds can be toxic to plants and can remain even after the soil has
75 dried out again (McKee and McKelvin, 1993). In more extreme cases when soils are subjected
76 to prolonged and complete submergence, the availability of CO₂, light and O₂ decrease,
77 severely reducing photosynthesis and respiration rates and ultimately leading to death in many
78 crop species (Jackson and Colmer, 2005) and a significant monetary loss to farmers
79 (Posthumus et al., 2009).

80 Soil chemistry can change considerably under waterlogged conditions leading to a
81 disruption in nutrient cycling (e.g., N, C and P) and excessive losses (Cabrera et al., 1999;
82 Sánchez-Rodríguez et al., 2017, 2018, 2019a, 2019b). Under anaerobic conditions, the N
83 mineralisation process is halted due to the lack of oxygen and as a result NH₄⁺ levels build up
84 to higher than normal concentrations (Unger et al., 2009). While NH₄⁺ is usually beneficial to
85 plants as a readily available form of N, in excess it can inhibit growth and even become toxic
86 to some plants (Loqué and von Wirén, 2004). Furthermore, pH can change when soils become
87 flooded (Ponnamperuma, 1972). If soil pH is altered sufficiently beyond the optimum levels
88 for plant growth, then the addition of lime or fertilisers may be necessary (Fernández and
89 Hoef, 2009).

90 Flooding can also cause physical changes to the soil (e.g. changes in soil structure and
91 bulk density), especially in fine clay soils (Jackson, 2004). Soil aggregate stability in the upper
92 layers reduces during long-term flooding as a result of several chemical processes, particularly
93 elevated pH, increased cation exchange and the prevalence of reduced conditions
94 (Ponnamperuma, 1972). This disaggregation and compaction of surface soils decreases the
95 chance of water draining away into the subsoil and increases the chance of surface capping,
96 which can hinder plant growth and soil drying once the floodwater recedes (Horn et al., 1995),
97 as well as increasing the risk of overland flow of water and pollutants.

98 Macrofaunal communities can survive short term flooding events (Zorn et al., 2005)
99 and can help alleviate some of the problems caused by flooding by burrowing to aerate the soil,
100 and transporting and releasing nutrients (Lavelle et al., 2006). However, although several
101 earthworm species can survive in aerated waterlogged conditions for some time (Zorn et al.,
102 2005), in anaerobic waterlogged conditions, macrofaunal communities can disappear due to
103 the lack of O₂ (Plum, 2005). Furthermore, soil microbial communities may change from a
104 diverse aerobic assemblage to a much less diverse and less active anaerobic community, which
105 can further contribute to changes in soil chemistry (Freeman et al., 2004).

106 To alleviate the effects of flooding on soils, the changes discussed above essentially
107 need to be reversed. Firstly, the soil needs to dry out, nutrients need to be restored and soil
108 structure needs to be improved to facilitate plant growth and further drainage and aeration of
109 the soil. On one hand, drying the soil is the crucial first step, and will remedy most of the
110 negative impacts of flooding (Ponnamperuma, 1984). On the other hand, if the soil is worked
111 by heavy machinery while it is still too wet, there is a risk that severe soil structural damage
112 can occur, especially in clay soils (Dexter and Bird, 2001). In particular, bulk density can
113 increase, water porosity decrease, aggregate stability decrease and the continuity of pores and
114 links to any drainage systems can be damaged (Dexter and Bird, 2001). To help improve
115 drainage, infiltration rates can be improved by reducing stocking density on grazed land to
116 minimise soil compaction (Castellano and Valone, 2007), planting cover crops to break up the
117 surface layers (Angers and Caron, 1998), introducing organic matter to the soil to improve soil
118 structure (Franzluebbers, 2002), or by cross field ploughing along contours rather than down
119 slopes (Puustinen et al., 2005).

120 Once the soils are sufficiently dry, heavier machinery can be used to break up the
121 compact soil (Spoor, 2006). Generally in wet soils, ploughing or sub soiling is often preferred
122 as the mechanical disturbance aerates the soil to a greater depth than other mechanical means

123 (generally >20 cm) (Strudley et al., 2008). Other cultivation methods include sward lifters,
124 which aerate the soil to a depth of 20 cm, or aerators, which aerate the soil to a depth of around
125 10 cm (Strudley et al., 2008). However, all of these cultivation methods require a tractor to pull
126 the equipment through the soil, which can cause compaction both on the surface and at plough
127 depth, depending on the furrows created by each method (Spoor, 2006; Strudley et al., 2008).
128 This can eventually result in a 'plough pan', which can then lead to further compaction and
129 reduced drainage in the future if the soil is not dry enough (Dexter and Bird, 2001).

130 Due to the rarity of extreme floods, relatively little is known of the long-term impacts
131 of prolonged inundation and subsequent recovery. Considering that we are predicted to
132 experience more extreme flood events in the future (Slingo et al., 2014), it is imperative that
133 we understand these impacts and, more importantly, how to mitigate and alleviate the damage
134 they might cause. The main objectives of our study were therefore: (1) to assess the effect of
135 the extreme UK winter flooding event (2013-2014) on physical, chemical and biological soil
136 quality indicators at 15 flood-affected sites; (2) to determine if these changes in soil health are
137 reversible in the short term (around 1 year), and (3) to determine the best methods for
138 alleviating flood damage caused by extreme winter flooding at 2 of these sites (sward lifting,
139 sub soiling, slot seeding and aeration in comparison with the control plots without
140 intervention). Our hypotheses were: (1) if the flood water column was considerable (0.3 to 1
141 m), it is possible that this would have a profoundly different impact on plant production, soil
142 biological, physical and chemical properties in comparison with a <0.3 m water column or
143 waterlogged soils; (2) if this water remains for an extended period, as it did in winter 2013-14,
144 perhaps even flood-tolerant crops may not be able to recover in the long term (a few months to
145 one year).

146

147 **2. Materials and methods**

148 *2.1 Study sites, experimental design, treatments and sampling timeframe*

149 Fifteen agricultural field sites were selected across Somerset, Worcestershire,
150 Herefordshire and North Wales to monitor the recovery of soils and vegetation after prolonged
151 flooding (Table 1, Sites 1-15). Sites were selected to cover a number of important agricultural
152 crops and soil types, and there needed to be clear evidence of unflooded and flooded areas at
153 the same site. Where it was possible (Sites 1 to 7 and 13 to 15), each site was divided into
154 ‘control’ areas that were those that had remained above the flood water and ‘flooded’ areas that
155 were those that had remained under water for long periods of time (8-12 weeks; Fig. A1). Initial
156 sampling took place in April 2014 (Sites 1 to 15; including floodwater samples, Table A1), just
157 after the last of the flood water had receded, and the final samples were taken eight months
158 later in December 2014. A subset of these sites with defined flooded and control areas (Sites
159 3, 4, 7, 14 and 15) were selected for a more detailed monitoring of soil recovery. Sampling was
160 carried out on these five sites every five weeks from the end of May 2014 through to the middle
161 of December 2014, resulting in a total of seven temporal sample points for each of these five
162 sites. In the meantime, these sites were managed (and fertilised) as usual according to the crop
163 grown at each one. At each site, three independent replicate plots (3 m × 3 m) were sampled
164 from the control or flooded areas. The same replicate plots were used for sampling throughout
165 the study. Aboveground biomass, soil respiration rate, water infiltration rate, soil bulk density,
166 soil pH, electrical conductivity (EC) and soil nutrients (available-P, NO₃⁻ and NH₄⁺) were
167 determined (0-10 cm depth) at the five detailed monitoring sites for each time period. At the
168 remaining ten sites, all the above measurements were made in April 2014 and December 2014
169 with the exception of soil respiration and infiltration rate, and phospholipid derived fatty acids
170 (PLFAs) were evaluated as indicators of soil microbial biomass and community structure in
171 April 2014 only for sites 1 to 6 and 13 to 15.

172 Additionally, two grassland sites in the Somerset Levels (Site 12 and 16) where the
173 flooding was most extreme were selected for an amelioration experiment. Both of these sites
174 had been under water for the longest period of time (12 weeks with >1 m depth of floodwater;
175 Table 1). The experimental plots were set up 4 months after floodwater removal when the soil
176 had dried out enough to allow heavy machinery trafficking. All treatments were slot-seeded
177 except the control treatment and the experimental design at each site was identical and
178 comprised four blocks ($n = 4$) of each treatment (10 m wide, 25 m long) namely: (1) unamended
179 control, (2) sward-lifted, (3) sub-soiled, (4) aerated, and (5) slot-seeded only (called slot-
180 seeded). The fields were sampled 4 times over a 12-month period after the experiments were
181 initiated. The same replicate plots were used throughout the experiment. Aboveground
182 biomass, soil respiration rate, soil infiltration rate, soil bulk density, soil pH, electrical
183 conductivity (EC) and soil nutrients (available-P, NO_3^- and NH_4^+) were determined (0-10 cm
184 depth) at sampling time. In addition, foliar mineral element concentrations were determined
185 after harvesting the above-ground plant biomass from small plots (40 × 40 cm). Subsequently,
186 the samples were dried (80°C, 72 h), ground, ashed (450°C, 24 h), the ash dissolved in HCl
187 (Adrian, 1973) and the mineral content determined on a 700 Series ICP-OES (Agilent
188 Technologies Inc., Santa Clara, CA).

189 All treatments, except the control, were slot-seeded with *Lolium perenne* L. to re-
190 establish the pasture lost by flooding (AHDB, 2017a). The other interventions were chosen
191 based on their ability to penetrate the soil at different depths as follows (Fig. A2):

- 192 • *Sub-soiler* (Viceroy moledrainer-subsoiler; Browns Agricultural, Leighton Buzzard,
193 UK): the deepest treatment, penetrating to a depth of 30-36 cm. The sub-soiler consists
194 of two tines that dig deep ruts into the soil approximately 2.5 m apart.
- 195 • *Sward lifter* (Grassland Shakaerator; McConnel Limited, Ludlow UK): the mid
196 treatment, penetrating to a depth of 20-25 cm. The sward lifter consists of three tines

197 over a width of 2.5 m, preceded by a row of sharp disks to break up the surface soil and
198 followed by a roller to flatten the turf. The sward lifter also vibrates as it is pulled
199 through the soil.

200 • *Aerator* (Slitmaster Grassland Aerator; Browns Agricultural, Leighton Buzzard, UK):
201 the shallowest treatment, penetrating to a depth of 10-15 cm. The aerator consists of
202 several sharp points over a width of 3 m that roll over the surface of the soil creating
203 several small holes.

204 These three mechanical interventions were chosen based on expert advice from local
205 agronomists and national guidance (AHDB, 2016, 2017b).

206

207 2.2. Measurement of soil physical quality indicators

208 Stainless steel bulk density rings (100 cm³; Eijkelkamp Soil and Water, Giesbeek,
209 Netherlands) were used to take three intact cores (0-10 cm depth) from each flooded and control
210 plot. The samples were subsequently, weighed, dried (105°C, 16 h), reweighed and dry bulk
211 density and gravimetric moisture content calculated. Infiltration rates (ml min⁻¹) were
212 measured in the field using a Decagon Devices mini disk infiltrometer (METER Group Inc.,
213 Pullman, WA) and calculating the average infiltration rate over a 30 min measurement period.
214 The only exception to this was the last sampling in the amelioration trial when a single ring
215 infiltrometer was used (Bagarello and Sgroi, 2004).

216

217 2.3. Measurement of soil chemical quality indicators

218 Soil samples (0-10 cm depth) from each plot were sieved to 2 mm for analyses.
219 Deionised water (25 ml, 4 h) was used to extract 10 g of each soil sample and pH measured
220 using a Hanna pH probe and electrical conductivity (EC) with a Jenway 4520 conductivity
221 meter (Cole-Parmer Ltd, Stone, UK). Soil plant-available P was measured by extracting soil

222 with 0.5 M NaHCO₃ (pH 8.5; 1:5 w/v, 200 rev min⁻¹, 0.5 h; Horta and Torrent, 2007),
223 centrifuging the extracts (14,000 g, 15 min) and determination of P colorimetrically in the
224 supernatant was done according to Murphy and Riley (1952) on a Powerwave XS plate reader
225 (BioTek Instruments Inc., Winooski, VT). Soil NH₄⁺ and NO₃⁻ were measured by extracting 5
226 g of soil with 0.5 M K₂SO₄ (1:5 w/v, 200 rev min⁻¹, 1 h), centrifuging the extracts (14,000 g,
227 15 min) and colorimetric analysis of the supernatant according to Mulvaney (1996) and
228 Miranda et al. (2001) respectively using a Powerwave XS plate reader.

229

230 *2.4. Measurement of soil biological quality indicators*

231 To determine changes in soil microbial biomass and community structure, phospholipid
232 derived fatty acids (PLFAs) were determined on 25 g soil samples (previously sieved to 2 mm)
233 according to Bartelt-Ryser et al. (2005) for Sites 1-6 and 13-15 (*n* = 4 per condition and site)
234 immediately after the floodwater had receded (Apr. 2014). No PLFA samples were collected
235 from sites 7-12 because the whole field was flooded and there were no suitable control areas.
236 The soil was sieved to pass 2 mm and immediately frozen (-80°C). One-hundred twelve
237 different fatty acids were detected in the soil samples used for PLFAs but only 32 of them had
238 a concentration higher than 0.5 % of the total PLFAs. These thirty-two fatty acids, classified
239 per taxonomic group, were: (1) 14:0 iso, 15:0 iso, 15:0 anteiso, 16:0 iso, 17:0 iso, 18:0 iso,
240 17:0 anteiso, 15:1 iso ω₉c and 17:1 iso ω₉c used for Gram+ bacteria (Ratledge and Wilkinson,
241 1988; Kieft et al., 1994; Paul and Clark, 1996; Zelles, 1999; Olsson et al., 1999; Bartelt-Ryser
242 et al., 2005); (2) 16:1 ω₇c, 16:1 ω₉c, 17:1 ω₈c, 18:1 ω₅c, 18:1 ω₇c, 18:1 ω₉c, 17:0 cyclo ω₇c
243 and 19:0 cyclo ω₉c were used for Gram- bacteria (Kieft et al., 1994; Paul and Clark, 1996;
244 Zelles, 1999); (3) 16:0 10 methyl, 17:1 ω₇c 10 methyl, 18:0 10 methyl and 18:1 ω₇c 10 methyl
245 for actinomycetes (Zelles, 1999); (4) 15:0 DMA as biomarker for anaerobic bacteria; (5) 20:4
246 ω₆c for protozoa (only 0.34 % of the total PLFAs; Paul and Clark, 1996); 18:2 ω₆c for

247 saprotrophic fungi (Paul and Clark, 1996); (6) 16:1 ω 5c as biomarker for putative arbuscular
248 mycorrhizal fungi (Olson et al., 1999); and (7) 14:0, 15:0, 16:0, 17:0, 18:0, 20:0, 22:0, 24:0
249 were found but were not assigned to a specific taxonomic group (Ratledge and Wilkinson,
250 1988; Niklaus et al., 2003). Some PLFA ratios were calculated to assess alterations in the soil
251 microbial communities (protozoa/bacteria or predator/prey, Gram+/Gram⁻,
252 saturated/unsaturated fatty acids, mono/polyunsaturated fatty acids, and precursor/cyclo fatty
253 acids).

254 Above-ground plant biomass was measured in 40 cm × 40 cm independent replicate
255 quadrats at each site to determine differences in plant productivity between flooded and control
256 areas. After collection, the samples were dried (80 °C, 16 h) and their dry weight determined.
257 Earthworm numbers were quantified within a 20 × 20 × 20 cm volume of soil for each plot.
258 The soil was excavated, hand sorted and any earthworms present counted before being returned
259 to the plot. Soil respiration rate was measured at each plot using an EGM-4 infra-red gas
260 analyser (PP-Systems Ltd, Hitchin, UK).

261

262 2.3. Statistical analysis

263 Permutational multiple analyses of variances (PERMANOVAs) were used to determine
264 differences between conditions (flooded, control) and sites ($n = 15$) at the start and at the end
265 of the observational study. The data were square root transformed, Euclidean distance
266 dissimilarity matrices were calculated for each analysis and Partial Eta Squared effect sized
267 (η^2_p) were calculated for PERMANOVA results, where a small effect was defined as ≥ 0.0099 ,
268 a medium effect ≥ 0.0588 , and a large effect ≥ 0.1379 . 1-way ANOVAs were used to compare
269 the soil and aboveground parameters between flooded and control areas both at the start and at
270 the end of the study, including PLFAs (taxonomic groups and ratios at the start of the study
271 only). Principal component analysis (PCA) was used for PLFAs taxonomic groups to assess

272 alterations in the soil microbial communities. Additional PERMANOVAs were done for each
273 condition (flooded and control) with the factors time (start and end data) and site.

274 To identify seasonal changes in measured parameters at the 5 more intensively
275 monitored sites, mixed-design ANOVAs were conducted on the monthly data to determine any
276 significant differences between conditions (flooded and control areas) and over time (7
277 samplings). The same statistical analysis was used at each individual site.

278 The amelioration study data was analysed using PERMANOVA to determine
279 differences between sites, treatments and over time, and for each site separately to find
280 differences between treatments and sampling times. Additionally, 1-way ANOVAs were run
281 for each site and the four-time samplings to find significant differences between the five
282 treatments. An Analysis of Similarities (ANOSIM) was used to identify any significant
283 dissimilarities between treatments at the individual sites and months. As ANOSIM is a type of
284 regression analysis Pearson's *r* effect size was used instead of Partial Eta Squared, where a
285 small effect is defined as ≥ 0.1 , a medium effect is ≥ 0.3 and a large effect size is ≥ 0.5 . Tukey's
286 post hoc test was done to find differences between treatments when 1-way ANOVA was
287 significant.

288 When PERMANOVAs were used, pairwise tests were used to determine where any
289 statistical differences lay (flooded vs. control areas, between sampling times and treatments)
290 and additional PCAs were used to determine which factors explained most of the variation in
291 the data (we only showed the principal components with a Eigenvalue higher than 1.0 and that
292 explained more than 5% of the variance; for more details see "Appendix: Details of Statistical
293 Analysis and Results", termed "Appendix" from now). The statistical analyses were performed
294 using the statistical package SPSS software v22.0 (IBM Inc., Armonk, NY) and Primer-e
295 software v6.0 (Quest Research Limited, Auckland, New Zealand).

296

297 **3. Results**

298 *3.1. Impact of flooding and subsequent recovery at 15 sites*

299 At the start of the observational study, there were significant differences with large
300 effect sizes between conditions ($P(\text{perm}) = 0.027$, $\eta^2_p = 0.633$) and sites ($P(\text{perm}) = 0.001$, η^2_p
301 $= 0.903$). A PCA analysis showed that soil moisture, soil EC and soil NO_3^- were the main
302 factors explaining 93.0% of the variance in the data (Appendix, Page 1, Table A1, three
303 principal components). On the one hand, bulk density, soil pH and soil P were significantly
304 lower in the flooded areas in comparison to the control areas ($P = 0.027$, $P = 0.004$, and $P =$
305 0.034 , respectively; Table 2). In contrast, soil moisture and soil EC were significantly higher
306 for the flooded areas ($P < 0.001$ in both cases). By the end of the observational study, there
307 were no significant differences between conditions except for soil pH, where the same pattern
308 as at the first sampling was observed ($P = 0.023$, Table 2), although there were still significant
309 differences with large effect sizes between sites ($P(\text{perm}) = 0.001$, $\eta^2_p = 0.925$; Appendix,
310 Pages 1-2, and PCA in Table A2, three principal components that explained the 96.1% of the
311 variance).

312 As expected, flooded areas differed between the start and end of the study ($P(\text{perm}) =$
313 0.001 , $\eta^2_p = 0.621$), although there were also significant differences between sites ($P(\text{perm}) =$
314 0.001 , $\eta^2_p = 0.881$). These differences between sites were more evident when the crops were
315 different. A PCA showed that soil moisture and soil EC were the main factors explaining 87.5%
316 of the variance in the data (Appendix, Page 3, Table A3, two principal components). Similarly,
317 control areas also changed over time ($P(\text{perm}) = 0.001$, $\eta^2_p = 0.783$) and again showed
318 significant differences between sites ($P(\text{perm}) = 0.001$, $\eta^2_p = 0.882$). A PCA showed that soil
319 moisture, soil EC, soil P and soil NO_3^- were the main factors explaining 95.2% of the variance
320 in the data (Appendix, Pages 3-4, Table A4, three principal components). The fact that both

321 flooded and control areas differed between the start and end of the study suggests seasonal
322 variation.

323 The total PLFAs and the percentage of anaerobic bacteria were significantly higher
324 under flooded conditions than in the control areas ($P = 0.018$ and $P < 0.001$, respectively),
325 while the opposite occurred for the percentage of fungi ($P = 0.017$) in April 2014 (Table 3).
326 None of the calculated PLFA ratios were altered by flooding. The PCA showed that Gram+,
327 Gram-, protozoa and fungi were the main factors that explained 81.5% of the variance (Fig. 1,
328 only two principal components). After the extreme flood event (April 2014), the soil microbial
329 communities shifted from being related to higher percentages of fungi, putative arbuscular
330 mycorrhiza fungi and protozoa in control areas to higher percentages of Gram+ bacteria,
331 actinomycetes and anaerobic bacteria (Sites 1, 2, 3, 4, 6, 14 and 15) or Gram- bacteria (Sites
332 5 and 13; Fig. 1) in the flooded areas.

333

334 3.2. Monthly monitoring of soil recovery from flooding at five sites

335 In general, there were significant differences over time for all the monitored variables
336 (Appendix, Pages 4-6, Table A5 for a PCA). The main effect comparing between conditions
337 (flooded/control areas) was significant for infiltration rates ($P = 0.034$, $\eta^2_p = 0.202$), soil NH_4^+
338 ($P = 0.031$, $\eta^2_p = 0.207$), soil NO_3^- ($P = 0.003$, $\eta^2_p = 0.321$) and plant biomass ($P = 0.020$, η^2_p
339 $= 0.230$). However, there were significant interactions for bulk density ($P = 0.005$, $\eta^2_p = 0.404$),
340 infiltration rates ($P = 0.040$, $\eta^2_p = 0.328$), soil EC ($P = 0.039$, $\eta^2_p = 0.329$) and soil NO_3^- ($P =$
341 0.004 , $\eta^2_p = 0.411$).

342 Fig. 2 shows the time course of the soil physical properties for the five sites. The winter
343 flood event produced an increase in the soil moisture until the end of the experiment in the
344 flooded areas in comparison with the control areas but the differences were only significant for
345 the sampling in September/October ($P = 0.039$; Fig. 2a). Bulk density (Fig. 2b) and infiltration

346 rate (Fig. 2c) were not altered by flooding but there were significant differences between
347 months for the control (August vs. September/October sampling for bulk density, $P = 0.023$;
348 July vs. August, $P = 0.019$, and September/October vs. November, $P = 0.020$, for the infiltration
349 rate) and the flooded areas (November vs. December, $P = 0.005$, for the infiltration rate). More
350 significant differences were found when looking at each site individually (Table 4). Soil
351 moisture was significantly higher in the flooded areas of the five sites for some specific months,
352 but bulk density and the infiltration rate were altered in contrasting patterns for the different
353 sites and even sampling times. Flooding reduced soil bulk density in Sites 7, 14 and 15 but it
354 was increased in Sites 3 and 4 (Table 4). Alterations in the infiltration rate of the flooded areas
355 did not follow a simple trend: for the flooded areas, it was increased at the beginning of the
356 recovery phase and later decreased in Sites 3 and 4, while it was increased at Site 14 and a non-
357 clear trend was observed at Sites 7 and 15 (Table 4).

358 Soil chemical indicators are shown in Fig. 3. Soil pH was significantly reduced in the
359 flooded areas (taking together the five sites) in July 2014 ($P = 0.031$). There was a significant
360 reduction in the soil pH between June and July for the control and the flooded areas ($P < 0.001$
361 in both cases) and an increase for the flooded areas between September/October and November
362 ($P = 0.035$; Fig. 3a). Looking at the flooded areas of each site individually, soil pH was
363 significantly higher in the flooded areas at Sites 3 and 15 (1 month for each site) and lower in
364 Sites 3, 4, 7 and 14 (1, 2, 4 and 2 months, respectively) in comparison with the control areas
365 (Table 4). A general increase was observed for soil EC of the flooded areas during the whole
366 sampling period and the five sites together, significantly for May ($P < 0.001$), June ($P < 0.013$)
367 and July ($P < 0.011$, Fig. 3b), although some decreases were observed for Sites 4 and 7 (Table
368 4). Soil EC was significantly reduced between May and June ($P < 0.030$), September/October
369 and November ($P = 0.025$) and increased between July and August ($P < 0.001$), and August
370 and September/October ($P = 0.021$, Fig. 3b).

371 For the five sites together, there were no significant differences for soil P, soil NH₄⁺ or
372 NO₃⁻ between the flooded and the control areas (Fig. 3 cde). The differences were more
373 associated with the sampling time: there was a reduction of the soil P in the control areas in
374 June vs. July ($P = 0.005$). A significant increase in soil NH₄⁺ and NO₃⁻ was observed when
375 comparing July vs. August ($P = 0.023$ and $P < 0.001$, respectively) and in soil NH₄⁺ in August
376 vs. September/October ($P = 0.050$ and $P < 0.001$, respectively) in the control areas, and in soil
377 NH₄⁺ ($P = 0.004$) in August vs. September/October and in soil NO₃⁻ ($P = 0.036$) when
378 comparing July vs. August in the flooded areas. In addition, a significant reduction in soil NH₄⁺
379 and soil NO₃⁻ occurred between September/October and November for the control ($P = 0.012$
380 and $P < 0.023$, respectively) and flooded ($P = 0.001$ and $P < 0.001$, respectively) areas. For
381 each site (Table 4), soil P was significantly reduced in the flooded areas except in Site 15 (no
382 significant differences), soil NH₄⁺ was increased in Sites 3, 4 and 7 (two, two and one months,
383 respectively) but decreased in Sites 14 and 15 (one and two months, respectively) in the flooded
384 areas. Soil NO₃⁻ increased in Sites 3, 4, 7 and 14 (one, one, two and one month, respectively)
385 but also reduced later in two of them, 4 and 7 (two and one months, respectively) in the flooded
386 areas.

387 A clear negative effect was observed for plant biomass in May ($P = 0.004$), June ($P =$
388 0.004) and July ($P = 0.005$) in the flooded areas, and then, the production was significantly
389 reduced between July and August for the control areas only ($P < 0.001$; Fig. 4a) because they
390 were harvested. This is in line with what happened individually in Sites 3 (increased in May
391 and quickly decreased in June), 4 and 14 but not with Site 7, where a positive effect of flooding
392 was observed for plant production (Table 4). A negative effect was also observed in the number
393 of earthworms and in the CO₂ flux in the flooded areas, with significant differences in August
394 ($P < 0.001$) and November ($P < 0.001$), respectively (Fig. 4bc). There were significant
395 differences in the number of earthworms between November and December for the flooded

396 areas (significant recovery of number of earthworms, $P = 0.015$) and for the CO₂ flux between
397 August and September/October for the control ($P = 0.016$) and the flooded ($P = 0.018$) areas
398 when we considered the five sites together. The lack of earthworms in the flooded areas of
399 Sites 3, 4 and 15 meant that no significant differences were found between conditions
400 individually (Table 4) in contrast with Sites 7 and 14. The effect of flooding in relation to the
401 CO₂ was negative for Sites 3, 4, 15 and 15 but then positive for Site 7 (Table 4).

402

403 *3.3 Mechanical interventions to promote amelioration of the soil after extreme flooding*

404 An overall analysis of both trial sites was conducted to find any overarching patterns,
405 however, there were no significant effects of treatment on soil indicators, although there were
406 significant differences between months ($P(\text{perm}) = 0.001$, $\eta^2_p = 0.750$) and sites ($P(\text{perm}) =$
407 0.001 , $\eta^2_p = 0.833$; Appendix, Page 7). A PCA showed that soil EC and soil P were the main
408 factors explaining 96.2% of the variation in the data (Appendix, Page 7, Table A6, two
409 principal components).

410 Looking at each site individually, Site 12 showed significant differences between
411 treatments ($P(\text{perm}) = 0.001$, $\eta^2_p = 0.166$) and months ($P(\text{perm}) = 0.001$, $\eta^2_p = 0.850$; Appendix,
412 Pages 7-8). A PCA showed that soil EC was the main factor explaining 93.9% of the variation
413 in the data (Appendix, Page 8, Table A7, one principal component). Then, Site 16 showed
414 significant differences between treatments ($P(\text{perm}) = 0.022$, $\eta^2_p = 0.127$) and months ($P(\text{perm})$
415 $= 0.001$, $\eta^2_p = 0.828$). A PCA showed that soil EC and soil P were the main factors explaining
416 97.0% of the variance in the data (Appendix, Page 9, Table A8, two main components).

417 Focusing on each site and time of sampling separately, a small number of significant
418 differences were found, although these contrasted between sites (Figs. 5, 6 and 7). Soil bulk
419 density was decreased when the aerator and the slot seeder ~~only~~ were used for Site 12 in August
420 2015 ($P = 0.027$), while for Site 16 bulk density increased in the order slot seeded \geq aerated =

Commented [DC1]: I am not sure what this means?

421 subsoiled = sward lifted \geq control treatment in October 2014 ($P = 0.025$) and slot seeder \geq
422 aerated \geq subsoiled = control treatment \geq sward lifted in August 2015 ($P = 0.025$; Fig. 5a).
423 Although no differences in infiltration rate were found for the different treatments, a large
424 increase was observed on the last sampling occasion (August 2015) in comparison with the
425 three first ones (Fig. 5b).

426 Soil pH was significantly reduced for the different treatments in relation with the control
427 plots (significantly only for aerated and slot seeded plots) in December 2014 ($P = 0.003$) and
428 February 2015 ($P < 0.001$) for Site 12, while the opposite occurred for Site 16 in three of the
429 four samplings ($P = 0.004$ in October 2014, $P = 0.050$ in February 2015, and $P = 0.002$ in
430 August 2015; Fig. 6a). The rest of the chemical indicators were significantly altered by the
431 different treatments just once for each of them (Figs. 6 bcd). Soil P and NH_4^+ concentrations
432 were reduced in the slot seeder plots in August 2015 for Site 12 only ($P = 0.016$) and in the
433 aerated plots in December 2014 for Site 16 ($P = 0.050$), respectively, in comparison with the
434 control plots (Fig. 6c). In December 2014, significantly higher concentrations of soil NO_3^-
435 were measured in the slot seeded plots than in the sward lifted and subsoiled plots for Site 12
436 ($P = 0.016$), and in the control plots than in the sward lifted plots for Site 16 ($P = 0.036$, Fig.
437 6d).

438 Not many significant differences were found in the biological soil properties (Fig. 7).
439 Significant differences between treatments were found only in February 2015 for the above-
440 ground plant biomass in the order slot seeded \geq sward lifted = control treatment = aerated \geq
441 subsoiled for Site 12 ($P = 0.043$, Fig. 5a). In the case of the CO_2 flux, we observed significant
442 differences in October 2014, with the control treatment plots emitting more CO_2 than the
443 aerated plots and then the rest of treatments ($P = 0.002$), and August 2015, when the control
444 plots were the ones emitting the minimum amount of CO_2 and the aerated plots the maximum,
445 for Site 16 ($P = 0.011$; Fig. 7c). Finally, some nutrient concentrations in the aboveground

446 biomass on each site were significantly higher in the grass grown on the subsoiled plots for
447 Sites 12 (N and Mg) and 16 (Ca) than in the grass grown on the control plots (Table 5).
448 Additional information is shown in the Appendix (Pages 10-11, Tables A9, A10)

449

450 **4. Discussion**

451 *4.1. Soil recovery assessment*

452 It is well established that the damage to crops and loss of soil quality under flooding is
453 dependent on various factors including: soil and crop type, duration of event (Jackson, 2004;
454 Jackson and Colmer, 2005), type of flooding (Sánchez-Rodríguez et al., 2018, 2019b), the
455 agricultural practices in the flooded area before the event (Sánchez-Rodríguez et al., 2017),
456 and the time when the event occurred (winter/spring/summer/autumn; Sánchez-Rodríguez et
457 al., 2019a). Some of these factors, such as crop type and agricultural practices related to them,
458 partly explains the variability in agroecosystem response observed between our sites (see also
459 Figs. A3, A4, A5, A6, A7, A8, A9, A10, A11). Our results also indicate how difficult is to
460 predict the effects of a prolonged flooding event on soil physical, chemical and biological
461 indicators. Here, we highlighted the importance of repeatedly monitoring a wide range of soil
462 quality indicators which may alter quickly over time (e.g. soil moisture, bulk density, pH, EC).
463 Despite this, it was difficult to identify consistent trends across the sites.

464

465 *4.1.1. Flood-induced changes in soil physical indicators*

466 Flooding may cause alterations in soil structure and induce compaction (Jackson,
467 2004). Contrary to expectation, however, soil bulk density was actually lower in the flooded
468 areas of the fifteen sites assessed in April 2014 and at three of the five sites evaluated monthly
469 in comparison with the non-flooded areas (decrease of 19%), however, this was only apparent
470 for Site 15 at the end of the monitoring period (December 2014). The lack of loss of soil

471 structure is consistent with no effect on soil water infiltration rate (Horton et al., 1994), as bulk
472 density was altered it is still possible that structure was affected by flooding. As we did not
473 directly measure structure or aggregate stability, further studies are required to critically
474 evaluate how they respond to flooding. The use of machinery to sow, fertilize, and aerate the
475 soil too quickly after floodwater removal (i.e. too wet) may also have contributed to more
476 isolated physical damage at some sites, for example in Sites 3 (spring onions, Fig. A4), 4
477 (swedes, Fig. A5) and 13 (grassland, Fig. A10) where soil erosion, more exposure of the roots
478 and a loss of soil structure were observed. In contrast to our study, severe degradation of soil
479 structure has been described in sites where the crop was either sown or harvested in autumn
480 and in newly established grasslands (Holman et al., 2003). Probably the most severe impact of
481 flooding occurs when the floodwater moves across the field in which case a complete loss of
482 topsoil can occur (Fig. A4, Fig. A10).

483

484 *4.1.2. Flood-induced changes in soil chemical indicators*

485 A good example of how difficult was to identify consistent trends across the sites was
486 pH, a key soil indicator that affects nutrient bioavailability and soil microbial communities.
487 After the floodwater had receded (April, 2014), the pH was significantly lower across the 15
488 test sites (Table 2, 0.4 units lower) and again in the monthly sampling (July 2014, 5 soils; Fig.
489 3a), but was increased for Sites 3 and 15 in one of the samplings (June and Nov. 2014,
490 respectively). The increase for acid soils such as Site 15 can be explained by the reduction of
491 Fe or Mn under anaerobic conditions and the pH decrease for the more alkaline soils due to
492 increased partial pressure of CO₂ (due to the lack of O₂) that promotes the production of H⁺
493 (for example Sites 3, 4, 7 and 14; Ponnampereuma, 1972). The rise in soil moisture observed in
494 the flooded areas after the flood event in the flooded areas was expected (94% higher in
495 comparison with the non-flooded areas), as was the increase in EC (104% higher) due to the

496 release of soluble salts from decaying vegetation and lack of plant demand. However, these
497 parameters are highly dependent on topography (soluble salts can be transported to places in
498 the landscape that are prone to being flooded) and whether the floodwater originated from
499 groundwater rise or overland flow.

500 Changes in soil conditions from aerobic to anaerobic under flooding and then back to
501 aerobic conditions, not only affects soil pH, but also nutrient dynamics and their bioavailability
502 (Figuereido et al., 2015). During flooding, adsorbed and occluded P may have been released
503 from the surfaces of Fe (Figs. A9, A11) and Mn minerals as they become progressively reduced
504 by the microbial community (Delgado and Torrent, 2000). In addition, P may be released from
505 senescing vegetation (Sánchez-Rodríguez et al., 2019b). While this P may be susceptible to
506 leaching, depending on the direction of water flow in the soil profile, it could also be re-sorbed
507 onto Al hydroxide surfaces or precipitated (Schärer et al., 2009). The initial decrease in P
508 bioavailability observed across our fifteen sites is consistent with a loss of P from the plant-
509 available pool (up to a 42% in comparison with the non-flooded areas) suggesting that extra P
510 fertiliser may be required to promote optimal crop growth.

511 In relation to available N in soil, no clear pattern emerged across the sites. The
512 significant increase in soil NH_4^+ measured in the flooded areas of Sites 3, 4 and 7 could be a
513 result of continued mineralisation of organic matter during the flood period combined with the
514 inhibition of nitrification due to the lack of O_2 (Unger et al., 2009). In addition, part of this soil
515 NH_4^+ and NO_3^- could have been immobilized by soil microorganisms or taken up by plants as
516 they started growing after floodwater removal. The transformation of NH_4^+ into NO_3^- by
517 nitrifiers, whose activity was inhibited during the flooding and partially during the soil recovery
518 (high soil moisture; Nielsen, 1996), could explain the increases in soil NO_3^- in the flooded
519 areas of Sites 3, 4, 7 and 14. Some sites received fertilizers during the soil recovery phase to

520 improve soil fertility for the next agricultural season, explaining the increase in soil EC and P
521 at the end of the monitoring period.

522

523 *4.1.3. Flood-induced changes in soil biological indicators and plant growth*

524 Plant biomass was negatively affected in the first few months after flooding, being
525 between 66 to 81% lower than in the control areas. Although Posthumus et al. (2009) and
526 Sánchez-Rodríguez et al. (2019a) showed how damaging summer floods can be on primary
527 production, our study exemplifies the destructive effect of a prolonged winter flooding,
528 especially when the crops are submerged for long periods. Nevertheless, this study also
529 highlights the importance of plant species. Overall, flooding decimated the spring onion,
530 swede and winter wheat crops while having no major effect on the grassland.

531 Our results showing a flooding-induced decline in earthworm populations are in general
532 agreement with Ivask et al. (2012). In that study, it was concluded that the loss of earthworms
533 under prolonged flooding indicated a loss of soil functionality. While we agree with this in the
534 short-term, our results strongly indicate that earthworm numbers recover within 1 year to those
535 seen in the unflooded controls. This implies that a loss of soil function is transitory if flood
536 events occur very infrequently (Coyle et al., 2017; Posthumus et al., 2009).

537 Soil respiration rates as well as microbial activity are good indicators of soil health, but
538 they are highly responsive to temperature and soil moisture and thus highly seasonal (Pendall
539 et al., 2004). Although we observed changes in microbial community structure and biomass,
540 this appeared to have little effect on soil respiration, indicating a high degree of functional
541 redundancy within the soil community. Despite this, the microbial biomass was 60% higher
542 after the floodwater had disappeared from the flooded areas in comparison with the unflooded
543 areas. We ascribe this microbial growth to the increased availability of labile carbon and
544 nutrients from the plant and microbial necromass formed during flooding. The increase in the

545 percentage of anaerobic bacteria and the reduction in fungal biomass (-28.4%) in comparison
546 with the non-flooded areas (mainly obligate aerobes) have been described previously under
547 prolonged flooding in a range of ecosystems (Freeman et al., 2004; Sánchez-Rodríguez et al.,
548 2017). Of note, is the loss of arbuscular mycorrhizal fungi which may have a long-term
549 negative impact on plant performance (particularly in low input systems) as well as potentially
550 affecting the crop's ability to withstand further stress events (Latef et al., 2016).

551

552 *4.2. Strategies to improve soil quality after prolonged flooding*

553 Overall, we observed few positive soil and sward responses to the four mechanical
554 interventions at our two trial sites. This was surprising given that these approaches are being
555 recommended to farmers to improve soil health in flood-affected areas (AHDB, 2016, 2017a).
556 In part, these recommendations are based on the assumption that flooding induces a loss of soil
557 structure and induces compaction, although this view is not supported by our multi-site study
558 (Fig. 2b). At both trial sites, soil bulk density was already low and no restrictions to root growth
559 are expected (i.e. $>1.4 \text{ g cm}^{-3}$). However, we did observe that the dead mat of vegetation and
560 thin layer of silt (ca. 3 mm deep) on the soil surface did appear to inhibit grass emergence and
561 prolonged anaerobic conditions at the soil surface, at least in the short-term (Fig. A4). The
562 aerator and slot seeding would have helped to break this surface layer. At Site 12, all four
563 treatments proved successful at lowering bulk density although this was best in the slot-seeding
564 only treatment which received minimal vehicle trafficking. At Site 16, however, the opposite
565 effect was observed. Based on visual inspection, we ascribe the increase in bulk density to
566 compaction induced by vehicle trafficking (e.g. compression along tyre tracks) clearly
567 illustrating that the response is site-specific.

568 Tillage operations to enhance soil aeration have been shown previously to reduce
569 earthworm density (Lees et al., 2016). Although earthworm numbers in the soil were very low

570 after flooding, there rate of recovery was not positively influenced by any of the interventions.
571 This is probably linked to the lack of observable response in many of the other soil quality
572 indicators and no increase in plant productivity, both of which are strongly linked to earthworm
573 abundance (Blakemore, 1997). In terms of plant growth, slot-seeding into the damaged sward
574 failed to promote greater biomass production, even though the plants visibly established. This
575 reflects our observations at other sites and from laboratory studies that older swards (Sites 12
576 and 16) are more resistant to winter flooding than newly established swards and can regenerate
577 relatively quickly (Sánchez-Rodríguez et al., 2019b).

578 Our results showed a different response to the four mechanical interventions at the two
579 sites. This is consistent with previous studies showing highly variable agronomic responses,
580 with both increases and decreases in soil quality and grass productivity reported (Bhogal et al.,
581 2011). These studies have suggested that mechanical soil loosening can be effective in
582 improving soil structure and increasing grass yields where soil compaction has been positively
583 identified and mechanical alleviation is effectively carried out. Where no compaction is
584 identified (as in our trials), it appears that while soil loosening improves soil physical
585 properties, it may reduce grass yield due to sward and root damage (Frost, 1988).
586 Consequently, we conclude that a pre-assessment of soil quality is undertaken before any
587 remedial work is undertaken after an extreme flooding, rather than relying on broad scale
588 agronomic guidance notes. Further work is also required to evaluate whether our treatments
589 would have caused a more positive impact if they had been applied at arable sites where soil
590 structure and compaction is typically greater.

591

592 **5. Conclusions**

593 Our field-based study clearly shows that extreme winter flooding can alter a range of
594 soil physical, chemical and biological indicators which may impact on the ability of soils to

595 deliver a range of ecosystem services. Primary productivity was heavily impacted in the winter-
596 sown arable cropping systems studied here, resulting in all cases to a loss of harvestable product
597 (between 0 and 19-34%). In contrast, much less of an effect of flooding was seen in the
598 grasslands, presumably as these perennials were better established and possess physiological
599 traits that make them more flood tolerant. Our data therefore lends support to the reduction in
600 arable cropping within high flood risk areas and a move towards land uses with greater soil
601 coverage (i.e. less erosion prone), more water storage capacity and which contain flood-tolerant
602 plants (e.g. grasslands, wetlands; Wang et al., 2012; Kharel et al., 2016). Our data also suggest
603 that more work is required to promote land restoration after extreme floods. The four
604 mechanical interventions trialled here showed little overall agronomic impact, however, these
605 options were based solely on government and industry guidance rather than on soil testing. In
606 some cases, basic soil testing would have proved beneficial to identify which soil properties
607 were sub-optimal, of which some can be easily rectified (e.g. pH) but others less so (e.g.
608 earthworms).

609 More studies like this are needed to better understand the different effects of extreme
610 flood events on agricultural production and soil quality with soil as a provider of ecosystem
611 services. It is difficult to predict extreme weather events and consequently studies such as ours
612 lack both in-field replication and field measurements prior to the event (i.e. preventing a robust
613 before-after-control-impact (BACI) design; Conner et al., 2016). Further, we lack
614 measurements of soil quality during the flood event itself. We therefore encourage more
615 replicated field experiments that can simulate prolonged flood events. In addition, it would be
616 useful to combine this with other common extreme events such as drought or ozone stress
617 which may occur at different times of the year (i.e. does flooding increase the severity of the
618 next stress event, or does it help build agroecosystem resilience?). It would also be beneficial

619 to gain a wider assessment of extreme flooding on soil functioning, including nutrient cycling,
620 the persistence of pests and diseases, greenhouse gas emissions and alterations in subsoils.

621

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630

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818 **Figure captions**

819 **Fig. 1** Changes in soil microbial community structure after an extreme flood event at 9
820 agricultural sites. Principal component analysis for the different taxonomic groups (based on
821 PLFAs) as a function of the sites ($n = 9$) and conditions (flooded and control areas) immediately
822 after floodwater removal (April, 2014). Principal component 1 vs. 2 (a), principal component
823 1 vs. 3, and the corresponding taxonomic groups for these subfigures (b and d). Symbols
824 represent the mean of four replicates per site and condition.

825

826 **Fig. 2** Temporal changes in soil physical properties after exposure to an extreme flood event.
827 Fifteen agricultural sites were monitored after the floodwater receded in April 2014. Values
828 represent means \pm SE ($n = 15$) for paired flooded and unflooded areas. The presence of
829 asterisk/s indicate significant differences (*: $P < 0.05$, **: $P < 0.01$, ***: $P < 0.001$) between
830 conditions.

831

832 **Fig. 3** Temporal changes in soil chemical properties after exposure to an extreme flood event.
833 Fifteen agricultural sites were monitored after the floodwater receded in April 2014. Values
834 represent means \pm SE ($n = 15$) for paired flooded and unflooded areas. The presence of
835 asterisk/s indicate significant differences (*: $P < 0.05$, **: $P < 0.01$, ***: $P < 0.001$) between
836 conditions.

837

838 **Fig. 4** Temporal changes in soil biological properties after exposure to an extreme flood event.
839 Fifteen agricultural sites were monitored after the floodwater receded in April 2014. Values

840 represent means \pm SE ($n = 15$) for paired flooded and unflooded areas. The presence of
841 asterisk/s indicate significant differences (*: $P < 0.05$, **: $P < 0.01$, ***: $P < 0.001$) between
842 conditions.

843

844 **Fig. 5** Effect of four different amelioration treatments (sward lifting, aeration, subsoiling and
845 slot-seeding) on soil physical properties at two grassland sites heavily impacted by an extreme
846 flood event. Time course (mean value and standard error; $n = 4$ per treatment) of soil physical
847 properties for the different treatments. The presence of different letters indicates significant
848 differences (*: $P < 0.05$, **: $P < 0.01$, ***: $P < 0.001$) between treatments.

849

850 **Fig. 6** Effect of four different amelioration treatments (sward lifting, aeration, subsoiling and
851 slot-seeding) on soil chemical properties at two grassland sites heavily impacted by an extreme
852 flood event. Time course (mean value and standard error; $n = 4$ per treatment) of soil physical
853 properties for the different treatments. The presence of different letters indicates significant
854 differences (*: $P < 0.05$, **: $P < 0.01$, ***: $P < 0.001$) between treatments.

855

856 **Fig. 7** Effect of four different amelioration treatments (sward lifting, aeration, subsoiling and
857 slot-seeding) on soil biological properties at two grassland sites heavily impacted by an extreme
858 flood event. Time course (mean value and standard error; $n = 4$ per treatment) of soil physical
859 properties for the different treatments. The presence of different letters indicates significant
860 differences (*: $P < 0.05$, **: $P < 0.01$, ***: $P < 0.001$) between treatments.