## This is the accepted manuscript version of the contribution published as:

Sünnemann, M., Siebert, J., **Reitz, T.**, **Schädler, M.**, **Yin, R.**, Eisenhauer, N. (2021): Combined effects of land-use type and climate change on soil microbial activity and invertebrate decomposer activity
*Agric. Ecosyst. Environ.* **318** , art. 107490

## The publisher's version is available at:

http://dx.doi.org/10.1016/j.agee.2021.107490

# *Combined effects of land-use type and climate change on soil microbial activity and invertebrate decomposer activity*

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Land use and climate change are two of the key forces driving soil organisms' activity and thus the ecosystem functions they provide. However, potential interactive effects of climate change and different land-use types on soil biological activity still remain unclear. Here, we studied soil biological activity in a large-scale field experiment initiated in 2014 in central Germany with two levels of input intensity (conventional *versus* organic treatment) and two climate scenarios (ambient climate *versus* "projected climate", i.e., increased temperature by +0.55°C and altered rainfall patterns across seasons). We measured soil microbial activity and invertebrate decomposer feeding activity across two years (2$^{rd}$ and 3$^{rd}$ year after establishment) in three-week intervals. Both soil biological activity measures were used as proxies for decomposition processes. Interactive effects of climate change and land-use types were not significant in the present study. Our results show that the projected climate reduced soil invertebrate decomposer activity by -16%, while soil microbial activity was not impaired. This suggests that even a slight increase in temperature together with a shift in precipitation patterns, can induce a significant reduction in soil functions like organic matter decomposition and nutrient cycling. Soil microbial (-9.6%) and invertebrate decomposer activity (-22%) were significantly lower in organic treatment compared to conventional treatment, which might be due to higher soil organic carbon and nutrient concentrations in conventional treatment in the short term. These findings highlight the need to

33 better understand the main drivers of short- and long-term effects on belowground functioning

34 to develop sustainable management strategies for healthy soils in a changing climate.


## 35 Keywords

36 Organic farming; conventional farming; climate change; soil invertebrate activity; soil microbial activity;

37 conversion to organic farming

# 1. Introduction

Climate change is one of the key driving forces threatening soil organisms and thus endangering functions and services of terrestrial ecosystems (Parmesan, 2006; Phillips et al., 2019). Climate models predict a rise of air temperatures by 1.0 to 3.7°C by the end of the century (Allen et al., 2018), accompanied by an increase in the frequency of extreme climate events, leading to changes in the availability of world's water resources (Huntington, 2006). At the same time, global food demand is rising rapidly due to a growing population, putting pressure on agriculture to increase yields and improve agricultural practices (Tilman et al., 2011).

Soils have a profound impact on ecosystem functions, including plant performance and agricultural production, thus linking them closely to human-wellbeing (Wall et al., 2015). Their highly heterogeneous systems include an extensive pore network that accommodates a large part of terrestrial biodiversity (Decaëns, 2010). The complex activities of these diverse organisms enable a wide range of ecosystem functions, on which the provision of ecosystem services depends, such as food production (Bardgett and Van Der Putten, 2014; Decaëns, 2010). However, intensive management practices as well as climate change currently put considerable stress on some soil organisms, thereby threatening their functional role (Amundson et al., 2015; Beaumelle et al., 2020). Contrarily, poor knowledge of potential interactive effects between these two

pervasive global change drivers limits our ability to predict alterations of belowground communities and functions.

As soil organisms are dependent on soil moisture (Decaëns, 2010), drought periods reduce their activity and abundance (Frampton et al., 2000; Kardol et al., 2011; Orchard and Cook, 1983; Riutta et al., 2016). In addition, the strength of drought effects depends on changes in temperature, as high temperatures can intensify detrimental drought effects on the activity of soil microorganisms (Butenschoen et al., 2011) and invertebrates (Thakur et al., 2018). At the same time, soil organisms are temperature-sensitive. For example, increased temperatures accelerate microbial metabolism at sufficiently high soil moisture levels, thereby enhancing microbial respiration and thus intensifying greenhouse effects (Crowther et al., 2015; Hanson et al., 2000; Liang et al., 2019; Wan et al., 2007), as soil carbon dynamics are largely dependent on microbial activity (Liang et al., 2017). Soil microbial activity, in turn, is driven by inputs derived from plants, which are also largely climate-dependent (Kallenbach et al., 2016; Rudgers et al., 2018). On the other hand, the microbial metabolism may adapt to elevated temperatures, which would reduce climate feedback effects in the long run (Allison et al., 2010). Adding to these complex responses of soil organisms to climatic drivers, the direction and strength of climate change effects may depend on local site conditions, such as land use type and management intensity.

Half of the terrestrial surface is managed by farmers, with conventional farming accounting for the largest share, while organic farming is steadily increasing (Meemken and Qaim, 2018). More diverse crop rotations and limited use of chemical pesticides in organic farming often produce positive outcomes, such as higher biological activity, biomass, and species richness (Hole, 2005; Lori et al., 2017; Mäder et al., 2002). However, such positive effects often depend on additional site-specific conditions, landscape context, and management decisions (Bengtsson et al., 2005), While conventional practices such as pesticide application are known to have detrimental effects on soil organisms (FAO et al., 2020), regular application of mineral fertiliser in particular is considered to have a beneficial short-term impact on the soil community, as it enhances biological activity through increased plant biomass production (DeMalach, 2018).

We address this challenge by investigating the effects of climate change in two management systems in the framework of the large-scale experimental research platform "Global Change Experimental Facility" in Germany (Schädler et al., 2019). To gain a comprehensive picture of soil biological activity responses to two environmental change drivers, we performed 27 samplings of soil invertebrate decomposer activity and 34 samplings of soil microbial activity across two years in a full-factorial combination of climate (ambient versus +0.55°C increase in temperature and shifts in annual precipitation patterns) and input intensity in two land-use systems (organic versus conventional management). We hypothesized that (1) climate change and (2) and the

organic treatment, mainly characterized by a lower fertilizer input intensity, will reduce soil invertebrate decomposer and microbial activity. Based on the assumption that fertilization will have a positive short-term effect on soil organisms (Treseder 2008), we expected them to be more resistant to climate extremes. Therefore, we hypothesized that (3) the impacts of climate change are more pronounced in organic than in conventional treatment in the short term. We do not claim that we can directly compare the effect sizes of both environmental change drivers – climate change and land use - and we expect that the timing of effects and underlying mechanisms differ (Yin et al., 2020). Moreover, the reader should note that our hypotheses refer to short-term effects of land use; the potential benefits of organic land-use treatment, in comparison to conventional treatment, for soil life may only materialize after several years.

# 2. Material and Methods

## 2.1 Research site

The research site is part of the field research station of the Helmholtz-Centre for Environmental Research UFZ in Bad Lauchstädt, Germany (51°22’60 N, 11°50’60 E, 118 m a.s.l.). Located within the central German dry area, climate conditions are characterized by a low annual precipitation (489 mm, mean 1896-2013) and an average temperature of 8.9°C (mean 1896-2013). These

106 semiarid conditions paired with carbonated loess substrates and a grassy vegetation led to the
107 development of a soil classified as a Haplic Chernozem. Hence, the area that had been maintained
108 as an arable field until 2012 is characterized by a highly fertile 40 cm deep humus layer
109 (Altermann et al., 2005).

110 ## 2.2 Experimental design

111 We took advantage of the Global Change Experimental Facility, a large field experiment
112 investigating the influence of a projected climate scenario on ecosystem processes within five
113 different land use types (for details on the experimental design, see Schädler et al., 2019).
114 Established in 2013, the experiment consists of 50 subplots (16 m x 24 m) arranged in 10 main
115 plots (Fig. S1). On half of the main plots, the climate has been manipulated according to projected
116 climate scenarios in the future (Fig. S1). These scenarios were informed by different regional
117 climate models for the year 2070-2100 (Döscher et al., 2002; Jacob and Podzun, 1997; Rockel et
118 al., 2008). While temperature is predicted to rise up to 2°C across seasons, the consensus model
119 estimates a reduction of rainfall in the summer months and an increase in spring and fall. The
120 climate manipulation is realized by a steel construction (5 m height) with automatically closing
121 roofs as well as a sprinkler system to simulate rainfall. The construction increases air temperature
122 by 0.55°C, reduces rainfall by 20% in summer and increases it by 10% in spring and fall,

123 respectively. However, the treatment did not increase soil temperatures significantly in the study
124 period (Fig. S3b). In the present study, for each season, a period of three months was defined by
125 adjusting the precipitation according to the projected scenario: spring (March-May), summer
126 (June-August), fall (September-November), and winter (December-February).

127 Within each main plot, five subplots represent five different land use types: conventional
128 farming, organic farming, intensively-used meadow, extensively-used meadow, and extensively-
129 used pasture. The subplots have been managed in accordance with regional agricultural
130 practices, including typical crop rotations. In our study, we focused on climate change effects on
131 conventional and organic farming systems, i.e. two out of the five land-use types (n=20 subplots;
132 2 land use types x 2 climate treatments x 5 replicates). In order to be managed according to local
133 recommendations, the two farming systems were subjected to a slightly differing crop rotation.
134 In the conventional treatment, a 3-year crop rotation of winter rape, winter wheat, and winter
135 barley was applied. Organic plots, on the other hand, were maintained according to the EU
136 regulation for organic agriculture based on a 6-year rotation (European Union 2007). Crop
137 rotation included winter wheat (year 2 and 5), winter barley (year 3 and 6), and - to ensure
138 nitrogen supply - legumes (alfalfa and white clover) every first and fourth year. Growing a legume
139 every three years is the minimum requirement of the EU guidelines for organic farming. However,
140 the aim of the GCEF was to bring crop rotation for organic treatment as close as possible to that

141 in conventional farming, in order to allow a direct comparison in 2 out of 3 years. A crop rotation
142 with 8-9 species would lead to a large number of possible combinations, each with specific
143 effects, which is why the crop rotation described above was chosen for the experimental design.
144 During the two-year period of our study, wheat (2015) and barley (2016) were grown in both
145 management types (Table S1). The conventional plots received regular applications of mineral
146 fertilizers and pesticides (herbicides, fungicides, and insecticides). On the contrary, according to
147 the recommendation for Chernozem soils in organic farming, no nitrogen fertilizer is applied in
148 organic plots. Nitrogen supply is only provided by nitrogen-enriching crops in the crop cycle every
149 third and sixth year, which are supplemented with rock phosphate and patent kali (K-Mg-S) (Table
150 S1). Since the use of herbicides is restricted in organic farming, weed control was done
151 mechanically (Table S2).

152 ## 2.3 Assessment of soil microbial activity and soil water content

153 Soil microbial activity was assessed by measuring soil microbial activity of soil samples under
154 controlled laboratory conditions using an $O_2$-microcompensation system (Scheu, 1992). The first
155 soil sampling took place in March 2015. Sampling campaigns were repeated every three weeks
156 year-round until April 2017. This resulted in a total of 34 soil sampling events, which allowed us
157 to draw conclusions regarding seasonal belowground patterns under projected climate

conditions. Soil samples were taken with a steel core sampler (1.5 cm in diameter; 15 cm deep) with 8 subsamples per plot to account for spatial heterogeneity. Samples were homogenized and sieved through 2 mm, stored at 4°C, and used to determine soil microbial respiration as a measure of soil microbial activity (µl $O_2$ $h^{-1}$ $g^{-1}$ soil dry weight) (Scheu, 1992b). Six grams of soil, which were previously acclimatised for 3 days in an airtight container at 20°C, were weighed into glass vessels, attached to the $O_2$-microcompensation system and measured all at once. Samples were measured in one of four machines, which was accounted for in the statistical analysis. In order to perform the analysis under constant conditions, the entire measuring part of the machine was immersed in a 20°C water bath during the 24-h measurement, as done before (Eisenhauer et al., 2010; Siebert et al., 2019; Thakur et al., 2018). Given the constant temperature conditions during the measurement, the variable may be regarded as "potential" activity, although other drivers of soil microbial activity varied according to the treatments, such as soil water and nutrient contents. Afterwards, samples were dried for three days at 75°C and weighed to determine soil water content.

### 2.4 Assessment of invertebrate decomposer feeding activity

We measured in situ soil invertebrate decomposer feeding activity responses to the treatments by applying the bait lamina test (Terra Protecta GmbH, Berlin, Germany). The method evaluates

175 the feeding activity of soil invertebrates and represents a rapid and standardized approach to
176 assess decomposition of organic materials in the top soil (Kratz, 1998). It has been successfully
177 used to study land use (Birkhofer et al., 2011), climate change (Thakur et al., 2018), and
178 interactive effects of grassland-use and climate change effects on soil invertebrate feeding
179 activity (Siebert et al., 2019). Nevertheless, the information on the main consumers of the bait
180 material differs among studies (Birkhofer et al., 2011; Eisenhauer et al., 2014) and may also
181 depend on detritivore community structure. As the feeding bait is composed of cellulose powder
182 (70%), wheat bran (27%), and activated carbon (3%), it is most likely that the test primarily
183 indicates the feeding activity of soil invertebrate decomposers. The test is carried out with rigid
184 PVC strips that have a series of 16 perforations with 1.5 mm diameter arranged in 5 mm distance.
185 The bait holes were repeatedly filled with the bait substrate and inserted into the soil in an
186 upright position with the upmost perforation just below the surface level. In order to avoid
187 damage during insertion, a steel knife was used to prepare the ground. To account for spatial
188 heterogeneity within the subplots, five strips were used per plot, as done previously (Siebert et
189 al., 2019; Thakur et al., 2018). At each sampling time point, the bait lamina strips were removed
190 from the soil after three weeks of exposure and directly evaluated in the field. Bait consumption
191 was rated as empty (1), partly empty (0.5), or filled (0). Consequently, soil invertebrate
192 decomposer feeding activity could range from 0 to 16 (maximum feeding activity). Mean bait

consumption per plot was calculated prior to statistical analyses by combining the data from the five strips per plot. The activity was assessed in the same three-week intervals as microbial activity measures. However, due to ongoing management activities related to tillage directly after each harvest, soil invertebrate decomposer feeding activity could only be determined at 27 points in time (Fig. S8).

### 2.5 Measurement of soil environmental properties

Soil abiotic parameters were analyzed in spring and fall 2015 and 2016. Soil samples were also taken with a steel core sampler (1.5 cm in diameter; 15 cm deep). To account for spatial heterogeneity, 8 samples per plot were taken and homogenized before measurements. Total organic carbon (TOC) content was determined by dry combustion using a Vario EL III C/H/N analyzer (Elementar, Hanau, Germany). Hot water extractable carbon (HWC) and nitrogen (HWN), which represent the labile organic C and N pools, were determined from 10 g of air-dried soil according to Schulz (2002). Briefly, soil/water suspensions (1:5 w/v) were boiled for 1 h under reflux. After cooling down to room temperature, 0.1 ml of 1 M $MgSO_4$ was added and the suspensions were centrifuged for 10 min at 6700 g. Supernatants were filtered (0.45 mm Minisart RC25 single-use syringe membrane filters, PP-housing, SartoriusAG, Göttingen, Germany) prior to the determination of hot water extractable C and N concentrations (mg kg-1), which was done

using an elemental analyzer for liquid samples (Multi N/C, Analytik Jena, Germany). Soil mineral nitrogen was extracted (20 mL 1 M KCl, 1.5 h) from 5 g of fresh soil and measured using flow injection analysis (FlAstar 5000, Foss GmbH, Rellingen, Germany). Plant available P and K were extracted from field-fresh soil using double lactate (1:50 w/v, pH 3.6, 1.5 hours; Riehm, 1942). Subsequently, P and K concentrations in filtered soil extracts were quantified colorimetrically using the molybdenum blue method (Murphy and Riley, 1962) and with a potassium-selective electrode (perfectION, Mettler-Toledo, Gießen, Germany), respectively.

### 2.6 Extraction and identification of soil decomposers

Soil decomposers were extracted in spring and fall 2015 and 2016, i.e. four times in total. Therefore, we took soil cores of 5 cm diameter for mesofauna (mites and Collembola) and 16 cm diameter for macrofauna (earthworms). For mesofauna, the extracted soil cores were heated from above in a Macfayden extractor for ten days and slightly cooled from below. The warming conditions and decreasing water availability force soil arthropods to move towards the cooler and wetter area. Here, they were collected by a vessel and preserved in a glycol water suspension. The same procedure was applied to extract macrofauna, while a Kempson extractor was used for this purpose (Kempson et al., 1963). After filtration, the samples were stored in 70% ethanol, sorted, and identified at order level (Mcfayden, 1961).

## 2.7 Statistical analyses

Linear mixed effects models were used to analyse the effects of climate, land use, season, and their interactions on soil microbial activity and soil invertebrate decomposer feeding activity. Year and sampling time as well as main plot served as random effects. To improve the model fit, soil invertebrate decomposer feeding activity was log-transformed prior to analyses. To analyse the treatment effects on soil microbial respiration, measuring device was added as an additional random term to the model to control for possible machine effects. Linear mixed effects models were calculated using the R-package "lme4" (Bates et al., 2014). Linear regression of microbial activity, invertebrate decomposer feeding activity and soil water content were performed with the lm() function. Data were tested for normality of residuals and homogeneity of variances using the Shapiro-Wilk and Levene's test. Pairwise comparisons were carried out to determine significant differences ($\alpha = 0.05$) between soil parameters in conventional and organic farming using the glht() function in the multcomp package. Principal component analysis was performed to identify microbial, faunal and environmental patterns, using the *factoextra* package. The PCA was performed separately for fall and spring (in each case combined for 2015 and 2016) to account for seasonal dynamics. The principal components were selected using the scree plot criterion. PCA analysis in fall was conducted excluding invertebrate decomposer feeding activity,

yield and potassium, as these properties were not unavailable for both samplings. All statistical analyses were conducted using R version 3.4.3 (R Core Team 2018).

246 # 3. Results

247 Soil microbial activity ranged from 0.3 to 2.5 µl $O_2$ $h^{-1}$ $g^{-1}$ soil dry weight with an average of 1.22
248 µl $O_2$ $h^{-1}$ $g^{-1}$ soil dry weight across all measurements. Microbial activity was not significantly
249 affected by the climate treatment, but by a significant interaction effect between land use and
250 season: conventionally managed plots had higher levels of soil microbial activity than organic
251 plots in summer, spring, and winter (Table 1; Fig. 1). Moreover, microbial activity revealed a
252 positive relationship with soil water content in both management types, especially in spring (Fig.
253 S5).

254 Soil invertebrate decomposer feeding activity per plot ranged from 0 to 50% of consumed bait
255 substrate. The activity was significantly influenced by main effects of climate and land use, but
256 no significant interaction effects were found (Table 1). The projected climate reduced
257 invertebrate decomposer feeding activity consistently across all seasons (-16%), while
258 conventional farming had higher levels of invertebrate decomposer feeding activity than organic
259 farming (+22%). In addition, soil invertebrate decomposer feeding activity was (marginally)
260 significantly affected by season and tended to be lowest in winter (Table 1; Fig. 1).

261 The PCA analysis of microbial activity, invertebrate decomposer feeding activity, soil
262 decomposers, and soil environmental properties revealed a pattern that distinctly distinguished

between conventional and organic plots, showing a significant effect of land use, especially in spring (Fig. 2a; Table S4). On the contrary, PCA analysis of microbial activity, soil decomposers and soil environmental properties in fall samplings showed no management-specific clustering, underlining the seasonal dependence of these patterns (Fig. 2b; Table S4). The interaction of climate and land use had a (marginally) significant effect on soil detritivores (Table S5). Current climate conditions lead to higher abundances in conventional plots, whereas organic treatment had a positive effect in the projected climate scenario (Fig. S7).

Despite similar dependency patterns in both forms of land use, a significantly higher content of total organic carbon, mineral N-inorganic, plant-available potassium, hot water extractable carbon, as well as hot water extractable nitrogen was found in conventional compared to organic treatment. However, soil water content, pH-value, and plant-available phosphorus did not differ significantly between the land use treatments (Fig. S6).

## 4. Discussion

We assessed the short-term effects of two of the most important global change drivers, climate change and input intensity in two land-use systems (Díaz et al., 2019; Schädler et al., 2019), on soil biological activity across two years in high temporal resolution. Our study shows two key results. First, we saw differing responses of the two groups of organisms to climate change: the

280 projected climate treatment reduced soil invertebrate decomposer activity significantly, while
281 soil microbial activity was not impaired. Second, in line with our assumption, the organic
282 treatment reduced soil biological activity after two years of organic management compared to
283 conventional management. This decreased soil biological activity on organic plots was
284 accompanied by lower concentrations of soil carbon and nutrients.

285 The observed decline of soil invertebrate decomposer activity under projected climatic
286 conditions is consistent with our hypothesis (1), stating that climate change constrains soil
287 biological activity. Soil invertebrate and microbial responses were decoupled in their response:
288 projected climate conditions did not affect microbial activity in a significant way. It is well
289 established that temperature increases biological rates (Brown et al., 2004) and thus promotes
290 soil organism activity (Allison et al., 2010; Lehmann and Kleber, 2015). However, as elevated
291 temperature also increases evapotranspiration (Dermody et al., 2007), this may reduce soil
292 moisture (Wan et al., 2002) and thus exacerbate water limitation, counteracting any positive
293 effect (Liu et al., 2009; Thakur et al., 2018).

294 Soil organism activity is strongly dependent on soil moisture (Coleman et al., 2004). However, the
295 ability of different groups of organisms to withstand drought varies considerably (Voroney,
296 2007). When soil water content decreases, so does biological activity (Riutta et al., 2016), with

297 invertebrates generally being more susceptible to stress than bacteria and fungi (Manzoni et al.,
298 2012). As the bait lamina test is mainly composed of cellulose, it is primarily a measure of
299 decomposers feeding activity. Particularly Collembola, Enchytraeidae, and earthworms appear
300 to be responsible for the largest share of bait consumption (Birkhofer et al., 2011; van Gestel et
301 al., 2003), but evidence is mixed (Eisenhauer et al., 2014). A decline in soil moisture leads to
302 reduced soil water films (Coleman et al., 2004), which increases soil hardness (Anh et al., 2014),
303 making movement more difficult, especially for larger soil fauna such as Enchytraeidae and
304 earthworms. In addition, drought disrupts faunal reproductive cycles (Maraldo et al., 2009) and
305 makes their food drier and hard to digest (Thakur et al., 2018). Therefore, drought severely
306 impairs soil invertebrates (Frampton et al., 2000; Maraldo and Holmstrup, 2010; Tsiafouli et al.,
307 2005; Wever et al., 2001), forcing them to migrate to deeper soil layers (Coyle et al., 2017), enter
308 diapause (Holmstrup, 2002), survive unsuitable conditions in a cocoon/egg stage (Fraser et al.,
309 2012), and/or causes mortality (Thakur et al., 2018) and thus can have significant effects on soil
310 community composition (Yin et al., 2019). At the same time, however, we observed higher
311 densities of detritivores in organic farming under projected climate conditions, suggesting that
312 organic farming methods might have positive effects in the long-term (Garratt et al., 2011). In
313 contrast to our assumption, soil invertebrate decomposer activity was not significantly positively
314 correlated with soil moisture (Fig. S4b), which may indicate that detritivore densities may not

315 necessarily reflect their impacts on certain soil processes. However, climate change may alter soil
316 properties slowly (Gelybó et al., 2018), underling the importance of future long-term studies that
317 explore the activity patterns and drivers of soil decomposer invertebrates in more detail, for
318 instance, by using monitoring systems that detect soil-dwelling microarthropods in real-time
319 (Dombos et al., 2017).

320 Contradicting our initial assumption, soil microbes did not seem to suffer from the summer
321 drought. This is surprising given that soil water content promoted microbial activity in our
322 experiment, which was also confirmed by others (Baldrian et al., 2010). However, resistance to
323 desiccation generally increases as organism size decreases (Lavelle and Spain, 2001), and many
324 groups of soil microbes are able to survive in extremely dry environments by employing a range
325 of strategies: regulating their internal water potential (Harris, 1981), undergoing anhydrobiosis
326 (García, 2011), or producing exopolysaccharides (Roberson and Firestone, 1992) enables them to
327 resist dehydration. At the same time, soil microbial community composition may have changed
328 under scenarios with reduced rainfall (Wallenstein and Hall, 2012), for instance, by being
329 increasingly dominated by drought-tolerant species and/or more persistent r-strategists (Schimel
330 et al., 2007).

331 In line with our hypothesis (2), the conventional treatment had a beneficial short-term effect on
332 soil organisms; their activity was higher in conventional plots compared to organic plots. Despite
333 the fact that organic practices aim to damp the environmental impact on ecosystems and
334 promote biodiversity (Luttikholt, 2007), biological activity was higher in conventional plots
335 compared to organic plots.

336 A main difference between the land-use treatments was mineral fertilization causing a rapid
337 increase of soil carbon and nitrogen concentrations in the conventional treatment, which in turn
338 led to higher yields in conventional plots (11.28 t/ha) than in organic plots (9.9 t/ha; mean values
339 over 2015 and 2016; unpublished results). Higher plant productivity is able to increase inputs of
340 organic materials (Geisseler and Scow, 2014), which was also shown for Chernozem soils (Wei et
341 al., 2008). The enhanced plant performance provides microbes with a higher amount of C sources
342 in form of root exudates, decaying roots, and aboveground residues, thereby boosting microbial
343 growth (Bais et al., 2006; Peart et al., 2001). Thus, mineral fertilization may have supported the
344 development of more active soil microbial communities and enhanced soil biological activity in
345 the present study. However, this short-term effect should be interpreted with caution, as the
346 long-term effects of mineral fertilization have been shown to cause detrimental effects on soil
347 organisms (Treseder, 2008) and negative climate feedbacks, both through $CO_2$ emissions during

348 production of mineral fertilizer and through $N_2O$ emissions occurring downstream (Jiang et al.,
349 2019) and are known to reduce soil carbon sequestration (Khan et al., 2007).

350 Nevertheless, nutrient addition appears to be a key component determining biological activity in
351 the present study. Organic practices avoid the use of mineral fertilizers, and for Chernozem soils
352 the guidelines only allow the inclusion of legumes in the cropping cycle for additional nutrient
353 input (European Union 2007). Legume-based systems improve several ecosystem properties
354 (Stagnari et al., 2017), including soil organic matter availability, as plant litter is its primary source
355 (Castellano et al., 2015; Jensen et al., 2012). However, green manures in particular, only show
356 positive effects if implemented over a sufficient number of years (Sacco et al., 2015). As our study
357 was conducted in the second and third year after the introduction of organic practices, soil
358 microbes have not yet been able to benefit from this important source of energy and nutrients
359 (Kuzyakov, 2010).

360 Similar to microbial activity, invertebrate decomposer feeding activity also increased under
361 conventional management. However, their activity did not appear to be directly linked to soil
362 properties, such as pH, C and N contents. Nevertheless, activity patterns of microbes and
363 invertebrates are generally thought to be tightly intertwined, as they interact in diverse direct
364 and indirect ways (Crowther et al., 2015). For example, by grazing, soil invertebrates alter

365 bacterial and fungal community composition and biomass (Crowther et al., 2011a, 2011b;
366 Dempsey et al., 2013). At the same time, invertebrates continuously fragment and displace litter,
367 thereby changing its availability as a microbial resource and its chemistry (Chang et al., 2017;
368 Filley et al., 2008; García-Palacios et al., 2013). Since these are processes that show their effects
369 mainly in the long term, future studies may have to explore the activity patterns of different taxa,
370 as a two-year study will not be able to reveal long-term consequences (Treseder, 2008).

371 In conclusion, our results indicate that the response of soil organisms to climate change was not
372 modulated by input intensity, but differed between soil microorganisms and soil invertebrates.
373 In contrast to invertebrate decomposers, soil microbial activity was not significantly affected by
374 climate change, illustrating that soil communities are highly divers and complex in their
375 responsiveness to changes in precipitation patterns (Siebert et al., 2019). Due to their short life
376 cycles, soil microbes may respond more quickly to environmental changes (Araújo et al., 2008),
377 resulting in rapid shifts of the community structure. As different microbial groups employ a range
378 of traits, this would have considerable consequences for the respective ecosystem. In this
379 context, it is necessary to extend the analysis beyond the measurement of microbial activity and
380 to take a closer look at changes in the microbial community structure (Kostin et al., 2020;
381 accepted Manuscript). At the same time, many ecosystem functions are only provided by the
382 joint activities of soil microbes and invertebrates (Simpson et al., 2012), and long-term studies

are needed to explore if treatment effects need more time to materialize. Notably, enhanced rainfall in spring and fall was not able to compensate for detrimental effects of summer drought on soil invertebrate decomposer activity, highlighting that even modest climate changes will exert considerable pressure on farming systems in the future. Long-term studies are urgently needed to gain a more comprehensive picture of land use-dependent climate effects on soil communities and processes. More detailed analyses of soil community composition will provide further insights into the consequences of co-occurring environmental change drivers and may facilitate the development of sustainable management approaches. Moreover, more research is needed on how changes in decomposition processes will alter soil fertility and crop yield under future conditions.

## Declaration of authorship

Author Contributions: MSch is part of the GCEF steering committee that developed the experimental platform. NE conceived the study on soil microbial and animal activity. JS, RY, and TR collected the data. MSü and JS analyzed the data. MSü, JS, and NE wrote the manuscript. All authors participated to the revision of the manuscript.

## Acknowledgements

We thank the staff of the Bad Lauchstädt Experimental Research Station (especially Ines Merbach and Konrad Kirsch) for their work in maintaining the plots and infrastructures of the Global Change Experimental Facility (GCEF), and Harald Auge, François Buscot, and Stefan Klotz for their role in setting up the GCEF. We also thank Alla Kavtea, Claudia Breitkreuz, Tom Künne, and Ulrich Pruschitzki for their support with lab and field work. We acknowledge comments by four anonymous reviewers that helped to improve the manuscript. MS, JS, and NE acknowledge support by the German Centre for Integrative Biodiversity Research Halle-Jena-Leipzig, funded by the German Research Foundation (FZT 118).

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