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Severe drought rather than cropping system determines litter decomposition in arable systems

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ABSTRACT

Litter decomposition is a fundamental process in soil carbon dynamics and nutrient turnover. However, litter decomposition in arable systems remains poorly explored, and it is unclear whether different management practices, such as organic farming, conservation agriculture can mitigate drought effects on litter decomposition.

Thus, we examined the effects of a severe experimental drought on litter decomposition in four cropping systems, i.e., organic vs. conventional farming, each with two levels of tillage (intensive vs. conservation tillage) in Switzerland. We incubated two types of standard litter (tea bags), i.e., high-quality green tea with a low C:N ratio and low-quality rooibos tea with a high C:N ratio. We assessed litter decomposition during the simulated drought and in the post-drought period during three years in three different crops, i.e., pea-barley, maize, and winter wheat. Subsequently, we assessed whether decomposition in the four cropping systems differed in its resistance and resilience to drought.

Drought had a major impact on litter decomposition and suppressed decomposition to a similar extent in all cropping systems. Both drought resistance and resilience of decomposition were largely independent of cropping systems. Drought more strongly reduced decomposition of the high-quality litter compared to the low-quality litter during drought conditions regarding the absolute change in mass remaining (12.3% vs. 6.5 %, respectively). However, the decomposition of high-quality litter showed a higher resilience, i.e., high-quality approached undisturbed decomposition levels faster than low-quality litter after drought. Soil nitrate availability was also strongly reduced by drought (by 32–86 %), indicating the strong reduction in nutrient availability and, most likely, microbial activity due to water shortage. In summary, our study suggests that severe drought has a much stronger impact on decomposition than cropping system indicating that it might not be possible to maintain decomposition under drought by the cropping system approaches we studied. Nevertheless, management options that improve litter quality, such as the use of legume crops with high N concentrations, may help to enhance the resilience of litter decomposition in drought-stressed crop fields.

1. Introduction

Litter decomposition, i.e., the breakdown and mineralization of plant organic matter, controls carbon cycling and nutrient supply in terrestrial ecosystems (Hättenschwiler et al., 2005), and is responsible for one of the largest carbon dioxide fluxes (Djukic et al., 2018). In agricultural systems, litter decomposition is tightly linked to soil respiration, soil carbon stability and thus to soil carbon stocks, which are the only carbon sinks in croplands and grasslands (Schmidt et al., 2011). Decomposition processes are primarily affected by climatic factors, e.g., temperature and precipitation (Steidinger et al., 2019), substrate quality, e.g. carbon to nitrogen ratio (Pei et al., 2019), and the composition and activity of the soil biological community (Glassman et al., 2018). Management, such as tillage and fertilization strategies, has a significant impact on litter substrate quality and factors influencing litter decomposition (e.g. soil microbial communities and nitrate availability; Knorr et al., 2005). However, the impact of such factors on decomposition is not always consistent. For example, nitrogen addition increases the decomposition

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of labile litter, but decreases the decomposition of litter with high lignin content (Finn et al., 2015). Therefore, changes in climate and agricultural management are likely to have a significant impact on litter decomposition, but the underlying mechanisms are often not well understood (Walter et al., 2013).

Drought is expected to become more frequent and severe in the future (Dai, 2013), posing a serious danger to agricultural productivity and ecosystem functioning (Rosenzweig et al., 2014). According to the climate scenarios for the Canton of Zurich, summer precipitation is projected to decrease up to 25 % by 2060 and up to 38 % by 2085 (NCCS, 2021). Drought has already now become a main factor retarding litter decomposition across European agroecosystems (da Silva et al., 2020). However, contrasting responses of decomposition after drought has ended have been reported, ranging from increased rates due to an accumulation of easily decomposable organic matter (Joos et al., 2010) to persistently reduced rates as a result of reduced microbial activity (Allison et al., 2013). The impact of drought on decomposition can be assessed in various ways. First, the resistance (i.e., the ability to withstand drought) can be tested (Orwin and Wardle, 2004). Furthermore, resilience (i.e., the ability to return to undisturbed conditions) can be examined after drought. However, the resistance and resilience concepts have rarely been applied to investigate how drought influences decomposition, considerably limiting our understanding of litter decomposition in croplands under climate change.

Field management strategies (cropping systems) are a main driver of litter decomposition in croplands, as for example tillage disturbs the soil and fertilizer increases nutrient availability (Aziz et al., 2013). Organic farming, as opposed to conventional farming, relies on organic fertilizers and reduces bare fallow periods, which improves soil organic carbon and potentially increases soil water storage (Tully and McAskill, 2020). In addition, increased soil biota and microbial abundance as well as activity under organic farming can lead to enhanced litter decomposition compared to conventional farming (Birkhofer et al., 2008; Spiegel et al., 2018). The effects of tillage on litter decomposition apply directly through the mechanical breakup of litter material into smaller pieces and mixing with soil, but importantly also indirectly by changing the activity of soil organisms, soil properties, and environmental factors such as temperature, gaseous concentration, nutrient availability, and soil water flow, both on the short but also on the long term (Bronick and Lal, 2005). Besides, labile soil organic carbon fractions are greater in organic farming systems with reduced tillage, which could potentially mitigate drought effects on decomposition (Bongiorno et al., 2019). However, little is known about the combined effects of organic farming and conservation tillage on litter decomposition and the mitigation of drought effects on decomposition.

Litter decomposition is traditionally measured using litter bags, filled with local or native litters. However, preparing uniform bags with litter allowing for detailed comparison between treatments and sites is difficult, thereby strongly restricting the number of study plots and replicates. Thus, Keuskamp et al. (2013) recommended using commercially available tea bags (green tea and rooibos tea) as standard litter. The observed changes in these teas are comparable to that of other (native) litter (Duddigan et al., 2020).

The general objective of this study was to investigate how litter decomposition is affected by drought in four different cropping systems, i.e., organic vs. conventional farming with two levels of tillage (intensive vs. conservation tillage). We assessed litter decomposition during the drought and during the post-drought period with two types of standard litter with distinct qualities (high vs. low C:N ratio), which were incubated in the topsoil of three crops (i.e., pea-barley, maize, and winter wheat). In addition, we assessed if drought-induced changes in decomposition are associated with changes in soil nitrate availability.

We tested the following hypotheses:

1) Cropping systems affect litter decomposition of both high- and low-quality litters, with higher decomposition in organic and conservation tillage systems.
2) Drought reduces decomposition of both high- and low-quality litters during drought and shortly after drought release.
3) Drought effects on and resilience of litter decomposition differ among cropping systems, with smaller drought effects and higher resilience in organic and conservation tillage systems.

2. Methods

2.1. Study site and soil management

Our study used the Farming System and Tillage experiment (FAST), aiming at investigating productivity and ecological impacts of the most important arable cropping systems in Switzerland and beyond (for more details, see supporting material). FAST is composed of two field experiments established next to each other. The FAST I experiment started in August 2009 with a 6-year crop rotation with winter wheat (year 1), maize (year 2), grain-legume crop (year 3), winter wheat (year 4) and a grass-clover mixture (years 5 and 6), while the FAST II experiment started one year later (2010), repeating the same crop rotation. Each experiment consists of conventional vs. organic farming with different levels of tillage (intensive tillage vs. conservation tillage, i.e., no or reduced tillage), resulting in four cropping systems, i.e., conventional intensive tillage (C-IT), conventional no-tillage (C-NT), organic intensive tillage (O-IT), and organic reduced tillage (O-RT). As herbicides are prohibited under organic farming, reduced tillage instead of no-till was applied to control weeds in the corresponding organic system. The four cropping systems were replicated four times in four blocks following a Latin-square block design, resulting in 16 main plots of 6 m × 30 m each. Our study was conducted in 2018 and 2019, during which a pea-barley mixture and winter wheat were grown in FAST I, and maize in FAST II. Sowing and harvest dates for respective crops are shown in Table S1.

The entire site was managed organically since 2002 (Wittwer et al., 2017). Since the start of the FAST trial, mineral fertilizers and herbicides were used regularly in the conventional systems, while the organic systems relied on organic fertilizers and mechanical weed control. More detailed information on fertilization of each crop is shown in supporting material. Intensive tillage (IT) was performed with a moldboard plough to a depth of 20 cm in both organic and conventional systems. In the no-tillage treatment (NT) of the conventional systems, the soil was not tilled but sprayed with herbicides since the beginning of the FAST trial in 2009. According to soil samplings in 2013 and 2014, several soil quality variables differed significantly among cropping systems, with promoted aggregate stability, soil biodiversity, and the abundance of macro-and microbiota (e.g. earthworms and arbuscular mycorrhizal fungi) under organic farming and RT or NT compared to conventional farming and IT. Furthermore, soil organic matter was not significantly different among cropping systems, but show a clear tendency with highest value under O-RT (1.44 %) against similar values under the three other cropping systems (1.38–1.39 %;Wittwer et al., 2021).

Volumetric soil water content (SWC) was continuously recorded at 10 cm depth in FAST I (pea-barley and winter wheat), with two replicates per cropping system (EC-5, Decagon Devices Inc., Pullman, WA, USA). For logistic reasons, SWC in FAST II (maize) was manually recorded at 20 cm depth at a weekly basis (EC-5, Decagon Devices Inc., Pullman, WA, USA) and the recorded values were corrected once with gravimetric water content of soil samples collected on 26 Sep 2018.

2.2. Drought treatment

The drought subplots were established in each main plot (16 main plots each in FAST I and FAST II), directly next to control subplots (each 5 m × 3 m in size), which received natural precipitation, reflecting a split-plot design within each main plot. We simulated periods of drought by excluding all precipitation with portable rain shelters during longer phases for the three crops in 2018 (FAST I and II) and 2019 (FAST I only). Rain shelters were tunnel-shaped, 5 m long, 3 m wide, and at the highest point approximately 2.1 m tall (for technical details see Hofer et al., 2016). Metal frames were covered with transparent and ultraviolet light-transmissible plastic foil (greenhouse foil, UV5, 200 μm, folitec Agrarfolien-Vertrieb, Germany). Rain shelters had a ventilation opening at the top along the full length, and were open at the bottom at all sides, allowing air circulation to prevent temperature increases. Rainwater running from the foils was collected in plastic pipes (PVC pipes, cut in half) and directed away from the rain shelters (about 2 m). In order to mimic the severe drought forecasted by the future climate scenarios for the Canton of Zurich (NCCS, 2021), we stimulated sufficiently long drought periods to achieve these sever conditions, resulting in 37 days, 71 days, and 55 days of drought treatment for pea-barley, maize, and winter wheat, respectively. The start date of the drought treatment was set to spring but varied for the three crops, due to the differences in crop growth seasonality and the timing of field operations, aiming at a strong but realistic drought period (Table S1).

2.3. Litter decomposition experiment

Following the standardized protocol by the TeaComposition initiative (Djukic et al., 2018) on assessing litter decomposition, we used two tea types with different C:N ratios, i.e., Lipton green tea (European Article Number, EAN no. 8 722700 055525) and Lipton rooibos tea (EAN no. 8 722700 188438), with nitrogen concentrations of 4 % vs. 1.2 % and C:N ratios of approximately 12 vs. 43, respectively (Keuskamp et al., 2013). Previous studies used tea bags of woven mesh of 0.25 mm (Keuskamp et al., 2013), which have been replaced by bags with a new nonwoven polypropylene mesh. However, there is no indication that the quality of the tea changed and thus we expect the nonwoven tea bags to be as suitable as the previously used woven tea bags (Mori et al., 2021).

In each crop, i.e., in 2018 (pea-barley, FAST I; maize, FAST II) and in 2019 (winter wheat, FAST I), three sets of tea bags were inserted vertically with a small shovel into the top 1–5 cm soil depth for three incubation periods. Each of the sets of tea bags contained 192 bags, with three pseudo-replicates per tea type per subplot. First, we buried two sets of tea bags right at the beginning of each drought treatment and retrieved one of these set at the end of the drought, called drought treatment (T1), to study the *drought resistance* (Fig. 1). After removing the rain shelters, we buried a third set of tea bags in the soil remaining until the end of our decomposition experiment (i.e., the respective harvests). Right before harvest, we retrieved both remaining sets of tea bags, those placed after the drought, called T2, to study the *drought legacy effect* and those placed at the beginning of the drought, called T3 (i.e., covering the drought and recovery period; Fig. 1). T3 samples were used to study how much of the drought effect on litter decomposition is compensated after rewetting until crop harvest (i.e., the *resilience*). Note

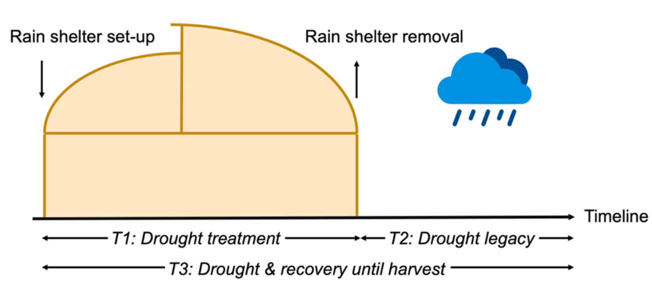

Fig. 1. Conceptual diagram of the drought treatment in relation to the timing of the litter decomposition experiment. Drought treatment and decomposition experiment were performed in pea-barley (FAST I) and maize (FAST II) in the year 2018, and in winter wheat (FAST I) in the year 2019. The sketch of the rain shelter (shown from short side) reveals the opening at the top and at all four sides to allow for intensive air circulation.

that dates of burial and retrieval of tea bags and corresponding dates of setting up and removing the rain shelters could not exactly identical for each crop due to logistic and especially crop growth as well as management reasons (Table S1). In total, we buried 1728 tea bags (3 pseudo-replicates × 2 tea types × 3 incubation periods × 4 cropping systems × 4 experimental blocks × 2 treatment subplots (drought, control) × 3 different crops). Our experimental approach with litter bags only considered the indirect effects of tillage on litter decomposition, such as changes in soil properties, soil water storage and soil microbial activities caused by tillage, rather than the direct effects of tillage on litter, i.e., mechanically breaking up litter material, as tea bags were incubated in soil after soil tillage (for details see Table S1). Note that the effect of crop species cannot be separated from other factors such as seasonal weather conditions, crop-specific management and differences in experimental phases. Thus, the study focuses on main drivers of litter decomposition, litter quality, drought and cropping systems. Therefore, we used each crop species as replicate for the statistical analysis aiming at the ability to generalize the results instead of analyzing the exact timing and duration of each treatment phase.

Before the start of all incubations, tea bags were oven-dried at 70 °C for 48 h and the initial weight was recorded. After retrieval, tea bags were cleaned from soil and roots and dried again at 70 °C until constant weight. We calculated the percentage of mass remaining (MR, %) by dividing the final dry weight of the tea bags by their initial dry weight. Heavily damaged or lost tea bags had to be excluded from further data analysis. This concerned in total 145 single bags (8 % of all bags buried). In one single case of missing data on subplot level, we replaced the MR with the average value from the same treatment of the corresponding plots to keep a balanced dataset for analysis.

2.4. Soil NO_3^- availability

Nitrate (NO_3^-) is the dominant form of available soil N in our system and a drought impacts on NO_3^- availability can help to confirm the treatment effect on soil nutrient fluxes and, indirectly, also on soil microbial activity (Marschner, 2012). To measure soil NO_3^- availability plant Root Simulator Probes (PRS Probes®, Western Ag Innovations, Saskatoon, Canada) for anions with a positively charged ion-exchange membrane were used. One sample consisted of four anion PRS probes per subplot as pseudo-replicates to cover spatial variability in the soil. PRS probes were inserted vertically in the soil, integrating over approximately 3–9 cm soil depth. For pea-barley, PRS probes were in the soil from 11 Jun to 26 Jun 2018 (15 days) to test the *drought effect* (T1), and from 26 Jun to 11 Jul 2018 (16 days) to test the *drought legacy effect* (T2) on the NO_3^- availability. In winter wheat, the *drought effect* (T1) was tested with PRS probes from 5 Jun to 17 Jun 2019 (12 days), while the *drought legacy effect* (T2) on available NO_3^- was tested between 17 Jun and 28 Jun 2019 (11 days). NO_3^- availability was not assessed for maize due to logistic and cost reasons. After retrieval, all probes were returned to the lab of Western Ag Innovations for NO_3^- analysis. There, NO_3^- was eluted from membranes with 17.5 mL of 0.5 mol/L HCl for one hour, and concentrations were determined colorimetrically using automated flow injection analysis (Qian and Schoenau, 2002). Ammonium concentrations have simultaneously been measured but were discarded because of extremely low values.

2.5. Crop yields

Crop yields of pea-barley and winter wheat were assessed by harvesting within two 0.25 m^2 areas that contain three rows of crops per plot by cutting the plants 1 cm above the ground (for further info see Table S1). Maize plants were harvested by cutting ten plants. Grain yield (t/ha of 100 % dry matter) was recorded after drying grains at 60 °C until constant weight and then calculated by transferring the values to tons per hectare.

2.6. Data analysis

Prior to analysis, the percentage values of litter mass remaining (MR) of the three pseudo-replicates per subplot were averaged. To understand the resilience of decomposition to drought, i.e., the ability to return to undisturbed conditions, we calculated the *resilience index* following the equation proposed by Orwin and Wardle (2004),

$$Resilience\ index = \frac{2|\Delta MR_{T1}|}{(|\Delta MR_{T1}| + |\Delta MR_{T3}|)} - 1 \quad (1)$$

where $|\Delta MR_{T1}|$ is the absolute difference of litter mass remaining between control and drought treatments at the end of drought treatment (T1) and $|\Delta MR_{T3}|$ is the absolute difference of litter mass remaining between control and drought at the end of decomposition experiment (T3). This index is standardized by the change caused by the drought treatment (MR_{T1}), which defines the state from which to recover. The index is bounded by − 1 and + 1, with values close to 1 indicating full recovery after rewetting, i.e., maximum resilience.

A repeated-measures ANOVA was conducted using linear mixed models (LME) to analyze litter decomposition for three incubation periods separately, using the R package "*lme4*" (Bates et al., 2015), accounting for the split-plot design of our experiment. Specifically, the effect of cropping systems (4 levels) was analyzed at the main plot level, the effect of drought (2 levels) at subplot level, and the effect of litter type (2 levels) at sub-subplot level; while "block", "crop", "cropping system (CS)", "drought treatment (D)" and "litter type (L)" were treated as fixed factors. Interactions were allowed for the latter three factors, i.e., CS × D × L. The factors "plot" and "subplot" were included as random terms. Linear mixed models were performed for each crop separately as well as for the three crops together to further investigate the effects of cropping system, drought treatment, and litter type. To unravel effects of cropping systems in absence of drought (hypothesis 1), models were run for control subplots only, with MR as response variable for the three incubation periods. Soil NO_3^- availability was analysed for fixed effects of CS and D using similar LMEs as described above. In case of significant drought treatment effects, post-hoc Tukey tests (at $\alpha = 0.05$) were used to test for pairwise differences among factors, using the R package "*multcomp*" (Hothorn et al., 2008). All analyses were performed with R version 4.0.2 (2020–06–22) (R Code Team, 2020).

3. Results

3.1. Environmental conditions and crop yields

3.1.1. Precipitation and air temperature

The year 2018 was comparably warm and dry, with an average annual air temperature of 11.3 °C, 1.5 °C above the long-term mean (1988–2017; Fig. S1), and a total precipitation of 853.8 mm, which was only 86 % of the long-term mean annual precipitation. Similarly, the mean annual air temperature in 2019 was 10.6 °C, about 0.8 °C above the long-term mean, while the annual precipitation of 1072.5 mm was 78.3 mm higher compared to the long-term mean (Fig. S1). During the period of the drought treatment (T1), the ambient mean air temperature was 18.7 °C for pea-barley, 20.3 °C for maize and 13.1 °C for winter wheat. The precipitation was 62 %, 61 % and 121 % of the respective long-term mean for pea-barley, maize and winter wheat respectively (Fig. S1). During drought recovery (T2), mean temperatures were 20.2 °C for pea-barley, 12.4 °C for maize and 20.7 °C for winter wheat. During the entire decomposition experiment (T3), the precipitation was 60 %, 54 % and 111 % of the long-term mean for pea-barley, maize and winter wheat, respectively (Fig. S1). Compared to the amount of precipitation received by the control subplots during T3, the drought treatments successfully excluded 64 % (pea-barley), 80 % (maize), and 69 % (winter wheat) of the precipitation (Figs. 2 and 3, Fig. S2).

3.1.2. Soil water content and soil temperature

In pea-barley, soil water contents (SWC) at 10 cm were similar in all subplots, before the start of drought treatment. After the rain shelters were set up, SWC in the drought subplots continuously decreased from around 25 % to around 15 %, before shelters were taken down. After removing shelters, there was no precipitation until 3 July (Fig. 2a), thus SWC stayed around 15 % until a sharp increase to an amount comparable to control subplots after the following precipitation (25.8 mm; Fig. 2a). In contrast, SWC in control subplots strongly varied with the incoming rain, typically between 20 % and 30 % (Fig. 2b). Soil

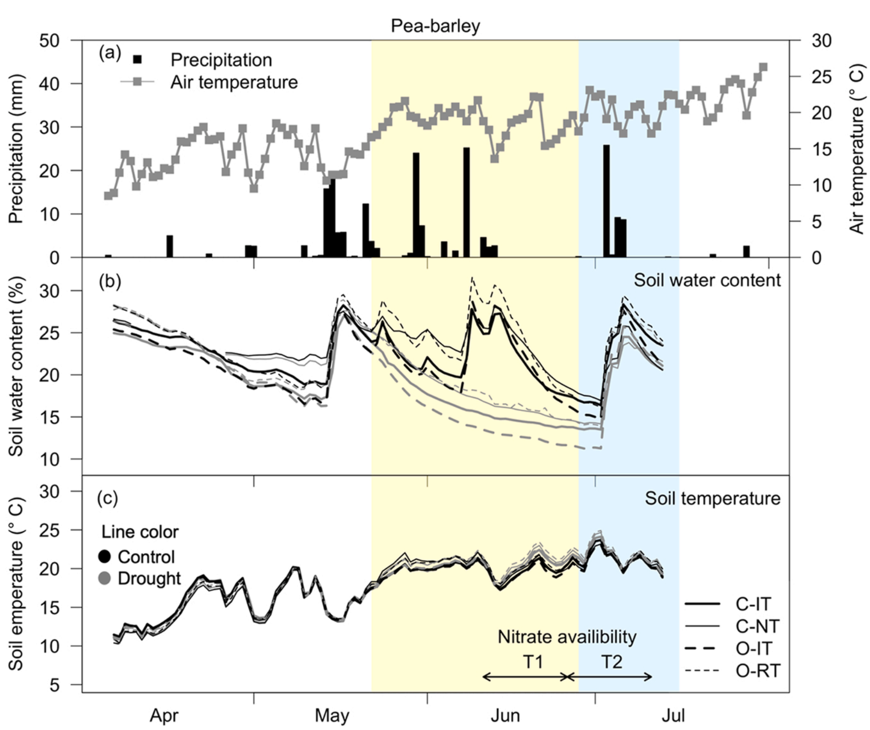

Fig. 2. Environmental conditions during the growing season in 2018. (a) Daily precipitation sums and average air temperatures; (b) daily mean soil water contents at 10 cm depth for the four different cropping systems (n = 2 each); (c) daily mean soil temperatures at 10 cm depth (n = 2 each). Shaded area in yellow represents the drought treatment period from 22 May to 28 June 2018; shaded area in blue indicates the post-drought period from 28 June to 16 July 2018 in pea-barley (FAST I). Timing and duration of measurement campaigns for soil NO_3 availability within the two phases T1 and T2 are indicated in panel (c). Abbreviations for cropping systems: C-IT: conventional intensive tillage, C-NT: conventional no-tillage, O-IT: organic intensive tillage, O-RT: organic reduced tillage. Lines in (b) and (c) use the same legend for cropping systems, with grey lines representing drought subplots and black lines representing control subplots. Climate data were obtained from a nearby MeteoSwiss weather station (Zürich/Kloten; see text for details).(For interpretation of the references to colour in this figure, the reader is referred to the web version of this article.)
Climate data were obtained from a nearby MeteoSwiss weather station (Zürich/Kloten; see text for details)

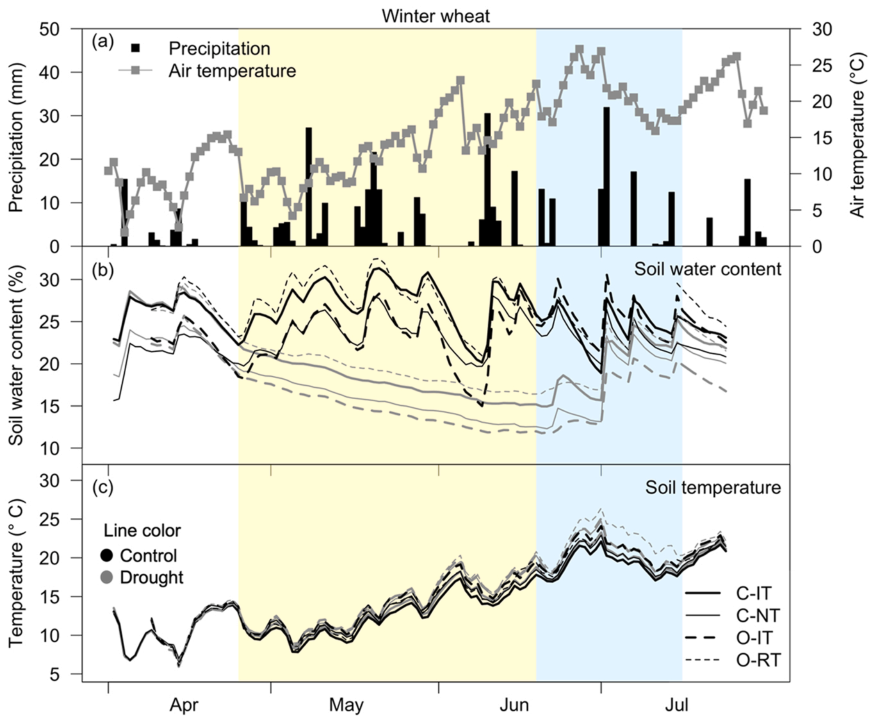

Fig. 3. Environmental conditions during the growing season 2019 for winter wheat. (a) Daily precipitation sums and average air temperatures; (b) daily mean soil water contents at 10 cm depth for the four different cropping systems (n = 2 each); (c) daily mean soil temperatures at 10 cm depth (n = 2 each). Shaded area in yellow represents the drought treatment period from 25 April to 19 June 2019; shaded area in blue indicates the post-drought period from 19 June to 16 July 2019 in winter wheat (FAST I). Timing and duration of the measurements campaigns for soil NO_3^- availability within the two phases T1 and T2 are indicated in panel (c). Abbreviations for cropping systems: C-IT: conventional intensive tillage, C-NT: conventional no-tillage, O-IT: organic intensive tillage, O-RT: organic reduced tillage. Lines in (b) and (c) use the same legend for cropping systems, with grey lines representing drought subplots and black lines representing control subplots. Climate data were obtained from a nearby MeteoSwiss weather station (Zürich/Kloten; see text for details).

temperature differences in all treatments were very small.

In maize, prior to drought treatment, SWC at 20 cm was similar in all treatments with around 23 % (Fig. S2). Afterwards, SWC decreased to around 13 % until in all subplots due to a natural drought. Starting from mid of August, SWC in control subplots began to increase again from 13 % to 23 % following a precipitation event. In drought subplots, SWC in conservation tillage systems (16 %) was relatively higher than the values in intensive tillage systems (11 %). After drought was relieved, SWC of C-NT and O-RT increased gradually to 22 % and 16 % respectively. At the same time, there was almost no increase in SWC in C-IT and O-IT despite the precipitation (Fig. S2).

In winter wheat, SWC at 10 cm was around 20 % before the drought treatment started (Fig. 3). SWC decreased during drought treatment to about 15 % in drought subplots but stayed high in control subplots (22–30 %). After removing the rain shelters, SWC in drought subplots increased slightly from 15 % to 18 % but remained much lower than in control subplots (25 %). This changed only on 1 July with a large precipitation event of 45 mm over two days, after which SWC in drought subplots increased to comparable values as in control subplots. Soil temperature differences in all treatments were small (during T1, on average, 0.6 °C warmer in drought vs. control subplots).

3.1.3. *Grain yields in control and drought subplots*

Drought reduced grain yields in all four cropping systems significantly by on average 20.3 %, 29.8 % and 15.6 % for pea-barley, maize and winter wheat, respectively (LME, $P < 0.001$, Fig. S3). Thus, the drought treatment was clearly effective in limiting water availability, even in case of maize where control plots temporarily exhibited a natural drought.

3.2. *Effects of litter quality and cropping systems on litter decomposition*

Over the two-year study period, litter decomposition was assessed for three different crops using two types of litter of different quality in each. Under natural conditions, i.e., in the control subplots, percentage litter mass remaining (MR) was significantly affected by litter quality, but—contrary to our expectation—not by cropping systems (Table 1). No interactions between litter quality and cropping systems were present. As expected, high-quality litter (green tea with a low C:N ratio) decomposed much faster than low-quality litter (rooibos tea with a high C:N ratio), with 50 % of MR for green tea and 81 % of MR for rooibos tea at the end of the decomposition experiment (average MR during T3 in control subplots of all the three crops; Fig. 4a). Litter decomposition significantly differed among the three crops, but as stated before, this variability also integrates the effects of weather and management during each individual crop growing season, as well as differences in timing and length of the respective incubation periods. Thus, we focus only on the results of the combined model using each crop as replication. However, the patterns found in each crop were largely consistent, e.g., regarding litter quality and cropping system effects (Table S2), so crop-specific

Table 1
Decomposition under ambient weather conditions: Effects of cropping systems (CS) and litter quality (L) and their interaction on percentage of litter mass remaining (MR, %) in control subplots during different incubation periods (T1, T2 and T3, respective dates for different crop growing seasons are shown in Table S1). *F* values and levels of significance given were derived from linear mixed model (LME). Please note crop species was used as fixed factor in the model but treated as replicates according to our experimental design. Results of single-crop models can be found in Table S2. Significant effects ($P < 0.05$) are indicated in bold.

		Drought treatment (T1)		Drought legacy (T2)		Drought & recovery (T3)	
All crops	df	*F* value	*P* value	*F* value	*P* value	*F* value	*P* value
Block	3	1.62	0.196	1.16	0.329	0.94	0.422
Cropping system (CS)	3	0.20	0.898	0.70	0.557	0.07	0.972
Litter quality (L)	1	**1864.90**	**< 0.001**	**718.18**	**< 0.001**	**1009.69**	**< 0.001**
Crop species	2	**15.56**	**< 0.001**	**11.23**	**< 0.001**	0.91	0.405
CS × L	3	0.05	0.983	0.40	0.754	0.31	0.820

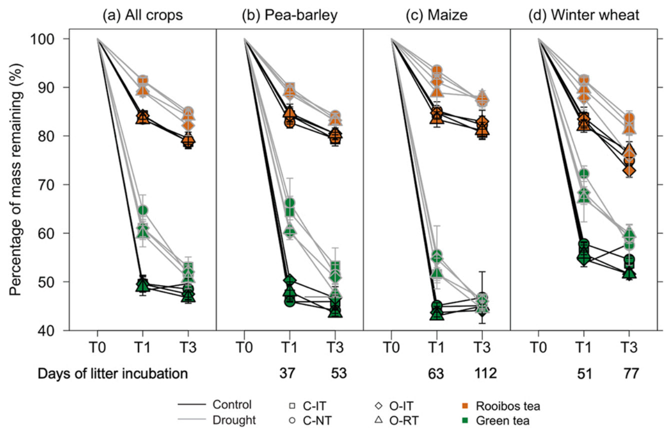

Fig. 4. Percentage of mass remaining (MR, %) for green tea and rooibos tea incubated in drought and control subplots over time, starting at T0 (start of litter exposure). Means and standard errors are plotted for T1 (end of drought, *drought effect*) and T3 (end of decomposition experiment, *drought residual effect*) in (a) all three crops (n = 12), (b) pea-barley mixture, (c) maize, and (d) winter wheat (n = 4 each) grown in different cropping systems. Days of litter incubation are given as well. Abbreviations for cropping systems: C-IT: conventional intensive tillage, C-NT: conventional no-tillage, O-IT: organic intensive tillage, O-RT: organic reduced tillage. Note that the lengths of the drought periods, the variability in weather and crop management for the three crops were different. See Table 2 and Table S3 for significant differences among drought treatments and cropping systems.

results highlight the robustness of the overall findings despite all potentially confounding factors.

3.3. Effect of drought on litter decomposition

3.3.1. Litter decomposition during drought

The drought treatment (incubation period T1) significantly reduced litter decomposition (all models with $P < 0.001$; Fig. 4, Table 2, Table S3). In contrast, cropping systems never had a significant effect on litter decomposition in the combine model including all three crops. There were also no significant interactions of drought treatment and cropping systems at the end of the drought (T1), indicating that the cropping systems studied were not able to mitigate drought effects on litter decomposition.

For all three crops and all cropping systems, ΔMR_{T1} values (i.e., the difference of MR between control and drought subplots (ΔMR) at T1) were positive and ranged from about 5–20% (Fig. 5a). As above, ΔMR_{T1} did not differ among cropping systems (Fig. 5a, Table 3, Table S4), but was significantly affected by litter quality (except for in maize; Table S4), with 12.3 % higher MR_{T1} for green tea and 6.5 % higher MR_{T1} for rooibos tea, averaged MR_{T1} for all three crops (Figs. 4,5a). Thus, litter quality interacted with drought effects, suggesting the decomposition of high-quality green tea was significantly less resistant against drought than decomposition of low-quality rooibos tea, independent of cropping systems. However, if expressed as a percent change of the average decomposition in control plots (100 - ΔMR_{T1}), decomposition of green tea decreased by 25 % and rooibos tea by 34 % due to drought. This shows that the change in decomposition of high-quality green tea was larger than the change in that of low-quality rooibos tea only in absolute but not in relative terms.

3.3.2. Resilience of decomposition after drought

After drought treatment and recovery period (T3), litter decomposition in drought subplots had not fully recovered, as the former drought treatment was still a highly significant factor, although drought was relieved two to four weeks ago (Fig. 4, Table 2, Table S3). Yet, note that the recovery period (T2) was much shorter than the drought treatment (T1). The effect of litter quality on decomposition was still significant ($P < 0.001$), while cropping systems still had no effect on litter decomposition ($P > 0.05$; Table 2). There were no interactive effects of drought × litter quality on ΔMR_{T2} in the combined model of the three crops (Table 2), but a significant interaction of drought × litter quality in maize ($P = 0.004$; Table S3).

Accordingly, the differences between drought and control subplots at the end of the experiment, i.e., ΔMR_{T3}, showed generally positive values for both litter qualities and all cropping systems, except for the decomposition of green tea under C-NT and O-RT in maize (Fig. 5c, Table S4). Note that these two subplots also showed higher SWC than C-IT and O-IT in maize after drought (Fig. S2).

To assess the ability of drought-affected decomposition to return to undisturbed conditions we further used the resilience index (Fig. 5d), which ranged from slightly below zero to slightly above 0.5. Full recovery would be found at one. No effect of cropping systems on resilience was found ($P > 0.05$), but litter quality again affected resilience ($P < 0.001$; Table 3). The decomposition of the high-quality litter had recovered much more than that of the low-quality litter ($P < 0.001$; Table S4), indicated by higher mean values of the resilience index for green tea (around 0.39) than that for rooibos tea (around 0.08; Fig. 5d).

3.3.3. Drought legacy effect on litter decomposition

After removal of the rain shelters until harvest (drought recovery: T2), decomposition of newly introduced litter was no longer affected by the previous drought treatment ($P = 0.121$) but only by litter quality ($P < 0.001$; Table 2). Similar to the pattern observed for the drought period, cropping systems did not show any effects on MR during T2 (all crops: $P = 0.258$, Table 2). The interaction of litter quality and drought was not significant after drought release ($P = 0.149$, Table 2). When crops were analysed separately, the main effects of the previous drought

Table 2
Decomposition of standard litters explained by the experimental factors. Effects of drought treatment (D), cropping systems (CS), litter quality (L) and their 2-and 3-way interactions on percentage litter mass remaining (MR, %) during different incubation periods (T1, T2 and T3, respective dates for different crop growing seasons are shown in Table S1). *F* values and levels of significance (*P* values) given were derived from linear mixed models (LME). Please note crop species was used as fixed factor in the model but treated as replicates according to our experimental design. Results of single-crop models can be found in Table S3. Significant effects ($P < 0.05$) are indicated in bold.

	df	Drought treatment (T1)		Drought legacy (T2)		Drought & recovery (T3)	
		F value	*P* value	*F* value	*P* value	*F* value	*P* value
Block	3	1.05	0.370	2.52	0.092	1.00	0.393
Drought treatment (D)	1	**167.04**	**< 0.001**	2.43	0.121	**33.67**	**< 0.001**
Cropping system (CS)	3	1.37	0.255	1.47	0.258	0.40	0.751
Litter quality (L)	1	**1886.92**	**< 0.001**	**1807.48**	**< 0.001**	**1674.94**	**< 0.001**
Crop species	2	**23.96**	**< 0.001**	**29.51**	**< 0.001**	**3.36**	**0.036**
D × CS	3	1.07	0.363	0.57	0.635	0.12	0.945
D × L	1	**15.90**	**< 0.001**	2.10	0.149	0.29	0.593
CS × L	3	0.42	0.738	0.81	0.498	0.50	0.683
D × CS × L	3	0.19	0.902	0.38	0.767	0.34	0.797

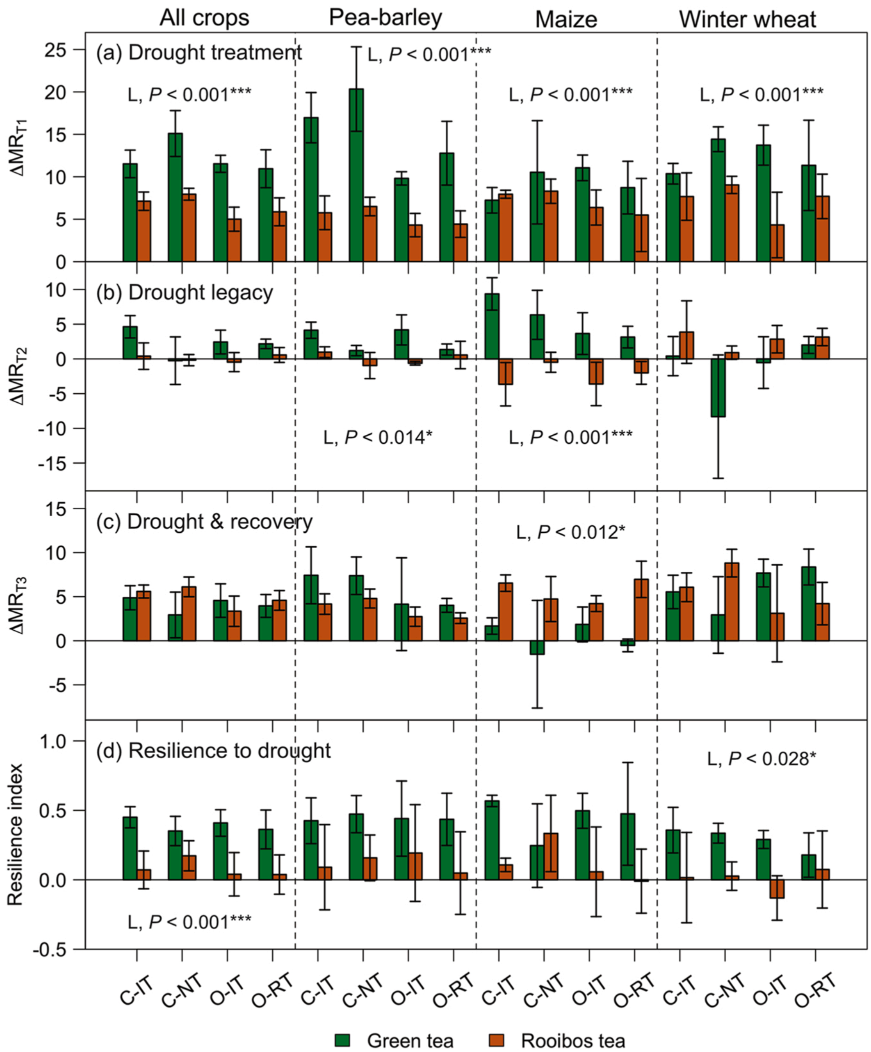

Fig. 5. Differences of mass remaining (ΔMR, %) in drought compared to control subplots for different litter quality (L), i.e., green tea and rooibos tea in four cropping systems (CS). (a) *Drought effect*, calculated as ΔMR for T1 (drought treatment), i.e., ΔMR_{T1}, with higher values indicating stronger effect; (b) *drought legacy effect*, calculated as ΔMR for T2 (drought legacy), i.e., ΔMR_{T2}, with higher values indicating stronger effect; and (c) *drought & recovery*, calculated as ΔMR for T3, i.e., ΔMR_{T3}, with higher values indicating worse *recovery after drought;* and (d) *resilience to drought*, based on the resilience index, with higher values indicating higher resilience. Means and standard errors are plotted for green tea and rooibos tea placed in different cropping systems for all three crops (n = 12) as well as in pea-barley, maize, winter wheat separately (n = 4 each). Definition of T1, T2 and T3 are given in Fig. 1. Abbreviations for cropping systems: C-IT: conventional intensive tillage, C-NT: conventional no-tillage, O-IT: organic intensive tillage, O-RT: organic reduced tillage. Significant effects of litter quality (L) and cropping systems (CS) are given in each panel (for more details on statistical models see Table S2). Levels of significance are given as * ($P < 0.05$), ** ($P < 0.01$) and *** ($P < 0.001$). Note that different lengths of the drought periods, the variability in weather condition and crop management for the three crops inhibit a comparison of crop species. See Table 3 and Table S4 for statistical models assessing the differences among treatment and cropping systems.

Table 3
Drought-induced changes in decomposition as affected by the experimental factors. Effects of cropping systems (CS) and litter quality (L) and their interaction on ΔMRT1, ΔMRT2, ΔMRT3, and resilience index. ΔMR (%) represents differences of litter mass remaining in drought compared to control subplots. Please note crop species was used as fixed factor in the model but treated as replicates according to our experimental design. Significant effects ($P < 0.05$) are indicated in bold.

		ΔMR_{T1}		ΔMR_{T2}		ΔMR_{T3}		Resilience index	
	df	*F* value	*P* value	*F* value	*P* value	*F* value	*P* value	*F* value	*P* value
Block	3	1.31	0.298	0.46	0.719	0.17	0.914	2.63	0.055
Cropping system (CS)	3	1.49	0.248	0.78	0.507	0.21	0.885	0.13	0.943
Litter quality (L)	1	**25.03**	**< 0.001**	2.89	0.093	0.57	0.452	**13.83**	**< 0.001**
Crop	2	0.96	0.390	0.25	0.778	2.08	0.138	1.23	0.296
CS × L	3	0.30	0.823	0.52	0.668	0.68	0.568	0.30	0.823

and the cropping systems as well as the interaction of drought with litter quality were not consistent among three outcomes (Table S3). Furthermore, but only for pea-barley, cropping systems significantly affected litter decomposition ($P = 0.004$; Table S2), but only for green tea with slightly higher decomposition in no/reduced tillage systems, i.e., C-NT and O-RT, than in the two intensive tillage systems (Fig. S4).

The differences in MR for the legacy period, so between control and previous drought subplots at the end of T2 (ΔMR_{T2}), was typically positive for green tea, reflecting the lower decomposition of high-quality litter in post-drought than in control subplots even after the drought was relieved (Fig. 5b). In contrast, ΔMR_{T2} of rooibos tea was relatively close to zero, suggesting similar decomposition of low-quality litter in post-drought and control subplots. Thus, ΔMR_{T2} was marginally significantly affected by litter quality ($P = 0.093$; Table 3, Fig. 5b). We did not find any effect of cropping systems on ΔMR_{T2} ($P > 0.05$; Table 3, Table S4, Fig. 5b).

3.4. Soil NO_3^- availability as affected by drought and cropping systems

As decomposition, nutrient availability and microbial activity are closely related, we studied the effects of drought and cropping systems on soil NO_3^- availability in pea-barley and winter wheat. In the unfertilized pea-barley, cropping systems had no effect on soil NO_3^-, while the drought treatment significantly affected soil NO_3^- both during and after drought (both $P < 0.01$; Table 4, Fig. 6a). During drought (T1), the soil NO_3^- availability in drought subplots was only 14 % of that in control

Table 4
Soil NO_3^- availability as affected by the experimental factors. Effects of drought treatment (D) and cropping systems (CS) and their interactions (D × CS) on soil NO_3 availability in pea-barley (2018) and winter wheat (2019) during (T1) and after (T2) drought (see Figs. 2 and 3 and Table S1 for exact dates of measurements). *F* values and levels of significance (*P* values) given were derived from linear mixed models (LME, n = 4). Significant effects ($P < 0.05$) are indicated in bold.

		Drought effect (T1)		After rewetting (T2)	
Pea-barley	df	*F* value	*P* value	*F* value	*P* value
Block	3	0.169	0.689	0.17	0.689
Drought treatment (D)	1	**39.69**	**< 0.001**	**20.69**	**< 0.001**
Cropping system (CS)	3	1.13	0.358	0.93	0.460
D × CS	3	0.52	0.673	0.96	0.440
Winter wheat					
Block	3	0.00	0.978	0.59	0.452
Drought treatment (D)	1	**11.61**	**0.005**	**11.60**	**0.002**
Cropping system (CS)	3	2.75	0.093	**5.81**	**0.004**
D × CS	3	0.93	0.458	0.77	0.521

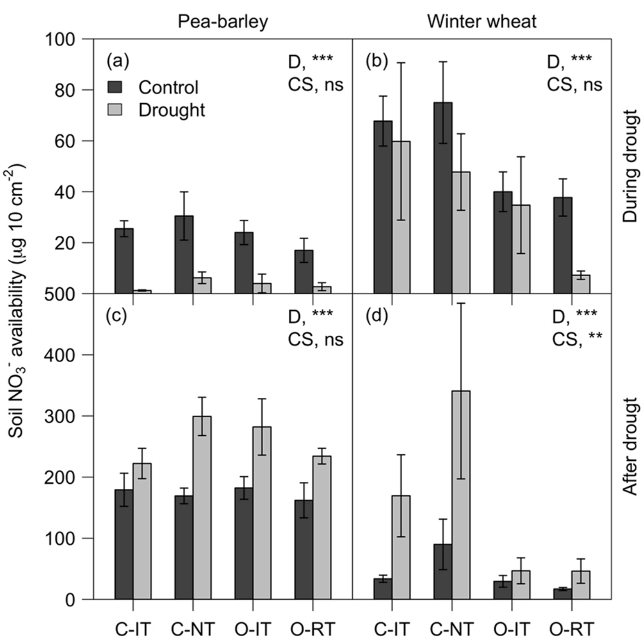

Fig. 6. Soil NO_3^- availability (mean ± 1 SE; n = 4) in control and drought subplots among four cropping systems (CS) for (a) pea-barley during drought, (b) winter wheat during drought, (c) pea-barley after drought, and (d) winter wheat after drought. Abbreviations for cropping systems: C-IT: conventional intensive tillage, C-NT: conventional no-tillage, O-IT: organic intensive tillage, O-RT: organic reduced tillage. Significant effects of drought treatment (D), cropping systems (CS), and their interaction (D × CS) are given in each panel (for more details on statistical models see Table 4). Levels of significance are given as * ($P < 0.05$), * * ($P < 0.01$) and * ** ($P < 0.001$).

subplots. However, after the drought was relieved (T2), NO_3^- availability increased particularly in the drought subplots, with up to 50 % higher values compared to control subplots. No interaction of drought and cropping systems was found for NO_3^- in pea-barely (Table 4, Fig. 6c).

In fertilized winter wheat, the drought treatment significantly reduced soil NO_3^- ($P = 0.005$; Table 4) for all cropping systems equally ($P > 0.01$; Fig. 6b). During this phase, cropping system showed no effects on soil NO_3^- ($P < 0.1$; Table 4, Fig. 6b). After rewetting (T2), soil NO_3^- availability increased in former drought subplots over concentrations in control plots ($P = 0.002$; Table 4), which was much stronger in the two conventional compared to the two organic systems (cropping system effect $P = 0.004$; Fig. 6d). As for pea-barley, no interaction of drought and cropping systems was observed for winter wheat (Table 4).

4. Discussion

Litter decomposition is an important soil function in croplands as it releases nutrients from organic matters to the atmosphere, to soil organisms, and crop plants (Spiegel et al., 2018). The understanding of how cropping systems and drought events affect litter decomposition is crucial to comprehend effects of global change on e.g. the carbon cycle, and it bears the potential to support strategies for climate change mitigation (Yin et al., 2019) and to inform Earth System Models (Bonan et al., 2013). In our study, we found a hierarchy in the experimental factors that drove the decomposition of standard litter in croplands. The strongest drivers were litter quality and severe (experimental) drought, while the cropping systems studied, i.e., organic vs. conventional farming with intensive or conservation tillage, had no or marginal effects on decomposition.

4.1. Litter quality but not cropping systems affected litter decomposition

Litter quality, reflected by N concentrations and C:N ratios of the litter, but not cropping system was the major determinant of litter decomposition in all incubation periods under ambient rainfall. Previous work supports the major relevance of litter quality (Martínez-García et al., 2021; Sanaullah et al., 2012), which is critical in determining litter decomposition when environmental factors such as temperature and moisture were accounted for (Shaw and Harte, 2001).

Contrary to our hypotheses, cropping systems had almost no effects on litter decomposition (Table 1). Only in one out of three crops (pea-barley), for one type of litter (high quality), and only for a short period after drought release–but not during the drought nor under control conditions for the entire experimental duration–cropping systems moderately affected litter decomposition (Table S2). Thus, in contrast to our expectation, neither organic farming nor conservation tillage consistently enhanced litter decomposition. The direct impacts of tillage such as breaking up and mixing plant litters and soil were most likely underestimated in our study, as no soil management was allowed after inserting the litter bags. However, indirect effects of conservation tillage on decomposition, which are known to potentially act via a higher water holding capacity and beneficial effects on soil organisms (Teasdale et al., 2007), were also not detected in our study. In previous work, such favorable effects of no-tillage vs. intensive tillage on the soil mesofauna were found to be considerably weaker compared to the effects of organic vs. conventional farming (Domínguez et al., 2014). However, under field conditions, Domínguez et al. (2014) reported higher decomposition in organic compared to conventional cropping systems with litter exposed for more than 4 months. In line with this, Martínez-García et al. (2021) argued that organic farming was able to enhance decomposition compared to conventional farming, but only when litter decomposed over more than two months. Although in our experiment, the incubation time for the litter bags was up to almost 4 months, we did not find such an effect.

Concerning conservation tillage, mass losses of litter were shown to linearly increase with the age of no-tillage (Houben et al., 2018). Earlier studies in the FAST trial we used for this study found that soil microbial communities did vary significantly with respect to organic farming and no tillage (Hartman et al., 2018) and also soil erosion, as an indicator of soil structural conditions, varied among cropping systems (Seitz et al., 2019). However, these changes in soil physical properties and microbial communities might still be too small to induce clear changes in litter decomposition that are attributable to the cropping systems.

In addition, as the mesh size of the litter (tea) bags did not allow larger soil organisms to enter, decomposition in our study can be mainly attributed to the activity of soil microbial communities and the mesofauna, but not to the macrofauna such as earthworms (Keuskamp et al., 2013; Sarneel et al., 2020). Thus, positive effects of organic farming and conservation tillage on earthworm abundances, as previously observed (Hole et al., 2005; Peigné et al., 2009), can be expected to have had a minor impact on litter decomposition in our study.

4.2. Strong drought effects on litter decomposition

The drought treatment excluded 64–80 % of the precipitation during the experimental duration, leading to considerably lower SWC (Figs. 2 and 3) and a significant reduction in crop yield by 16–30 % in drought compared to control subplots. In contrast, soil temperatures changed over time but did not differ among cropping systems and were only marginally affected by drought. Thus, our experimental set-up was very effective in creating severe drought conditions in all three crops, despite two unusually warm years (2018 and 2019; Fig. S1).

In line with our second hypothesis, drought strongly reduced litter decomposition for both litter qualities (Table 1). As outlined above, due to the litter bags used in this study, decomposition can be mainly attributed to the activity of soil microbial communities and the mesofauna (Sarneel et al., 2020). It is well known that the activity of soil microorganisms, and therefore also their effect on litter decomposition, is strongly driven by environmental factors such as soil temperature and soil moisture (Schlesinger and Bernhardt, 2013). As soil temperature differences during our drought treatment (T1) were negligible (drought vs. control + 0.4 °C for pea-barley and + 0.6 °C for winter wheat; no data available for maize), but SWC in the drought subplots continuously decreased, the highly significant drought effect on litter decomposition is most likely due to water shortage for soil biota. Such negative drought effects on soil biota are in agreement with previous observations of substantially reduced microbial activities under drought (Sanaullah et al., 2012; Walter et al., 2013). Both the abundance and diversity, and thus the functioning of microbial communities can be impaired by drought (Cavicchioli et al., 2019). In concert with the assumed reduced microbial activity, nutrient availability clearly decreased by drought in pea-barley and winter wheat, independent of cropping systems (Fig. 6).

Drought effects on decomposition were more pronounced for high-quality litter (green tea) than for low-quality litter (rooibos tea, T1; Table 1, Fig. 5), but only when the absolute but not the relative change in decomposition was considered. This is consistent with Sanaullah et al. (2012), stating that in a grassland the relative decomposition of low-quality litter was more affected by drought than high-quality litter. A possible explanation for this is the preference of microbial colonization for high-quality litter (Pei et al., 2019). As soon as microbial activities are impaired by any stress, decomposition of high-quality litter is more strongly affected than the decomposition of low-quality litters, and thus also shows larger immediate effects, i.e., lower resistance.

As the effect of the severe drought on litter decomposition did not differ among cropping systems and was widely consistent for all the three crops, our results indicate that the studied cropping systems could not mitigate the drought limitations of litter decomposition, independently of the question whether the crop was fertilized or unfertilized (but contained legumes). This finding clearly rejects our third hypothesis. As discussed above, although differences in soil properties have been caused by the cropping systems, this was potentially not strong enough to affect decomposing and its response to drought, as indicated by soil organic carbon differing among cropping systems only be tendency (Wittwer et al., 2021). However, as explained above, regarding tillage no direct but only indirect effects via changes in soil (ecological) properties could have been found, because the physical process of tillage was not applied to our litter bags. Our results are further in line with Diekötter et al. (2010), who found no differences in litter decomposition between organically and conventionally managed arable fields for wheat straw exposed less than 30 days.

Until now, only few studies assessed interactions of drought and crop management on litter decomposition. Da Silva et al. (2020) reported that drought equivalently reduced litter decomposition under conventional and organic farming in a greenhouse-based pot experiment. Likewise, Yin et al. (2019) found that increased drought and elevated temperatures decreased litter decomposition rates similarly for conventional and organic croplands under field conditions. Both studies are consistent with our results. However, in a greenhouse-based pot experiment, the use of organic fertilizers, as usually employed by organic farming, alleviated drought effects compared to pots with mineral fertilizer (Dimkpa et al., 2020). Thus, potentially beneficial effects of organic farming and conservation tillage on soil health and drought resistance might have occurred in a situation with a less severe drought. In this experiment, the long period of precipitation exclusion could have completely overrun such effects. Nevertheless, many regions in Europe have already experienced several extremely hot and dry summers, especially in 2003, 2015, and 2018, and droughts are projected to become increasingly more frequent and severe in the future (Spinoni et al., 2018).

The observed reduction in litter decomposition due to drought is clear evidence for a potential impact of climate change on carbon and nitrogen cycles. While such a change in nutrient cycling is per se concerning, as it might further impact other ecosystem processes and services, e.g., soil supporting services and crop growth, the ultimate impacts of reduced litter decomposition in arable ecosystems is not yet clear. On one hand, reduced decomposition can cause decreased availability and turnover of nutrients in arable farming system, which could limit the growth of plants that heavily rely on the recycling of nutrients like phosphorus and thus stimulate the addition of extra fertilizers by farmers (Schulze et al., 2019). On the other hand, when precipitation is limiting, reduced decomposition of litter may cause an accumulation of soil organic matter and subsequently increase e.g. water holding capacity (Bontti et al., 2009).

4.3. Resilience of decomposition after drought

When following the same set of litter incubated throughout the drought & recovery period (T3), we observed persistent and significantly lower decomposition in the post-drought subplots for all litter qualities compared to control subplots, confirming our second hypothesis. Yet, cropping systems did not affect the *resilience* of decomposition following the drought, which further rejects the third hypothesis (Table 1). Post-drought changes in mass remaining indicate that litter decomposition was catching up after drought was released, as can be seem from the decreased difference among drought and control samples at the end of the experiment compared to after the drought (Fig. 4). An explanation for this development is that litter decomposition is no linear process but slowed down over time the more recalcitrant the leftover material becomes (Hättenschwiler et al., 2005). This likely will have happened earlier in control compared to the delayed drought plots.

The resilience of decomposition significantly differed between the two types of litter. As discussed above, the decomposition of high-quality litter showed lower resistance to drought than the decomposition of low-quality litter, when looking at the absolute and not relative change. However, when assessing the *resilience* of decomposition after drought (Fig. 5d, Table S4), results suggest that high-quality litter showed a larger *resilience*, i.e., returning about halfway to undisturbed conditions, compared to low-quality litter. This observation can be explained by the decomposition rate of green tea being generally higher than rooibos tea, approaching a stable phase of litter decomposition considerably quicker than rooibos tea (Keuskamp et al., 2013). Thus, our results suggest that resistance and resilience of litter decomposition are negatively related, i.e., high-quality litter was associated with low resistance (higher vulnerability) but high resilience to drought, and vice versa for low-quality litter. Similarly, De Vries et al. (2012) found drought resistance and resilience of soil food webs, which govern litter decomposition, to be negatively related, with resistance being higher and resilience being lower in an extensively managed grassland (with fungal-based food webs) than that in an intensively managed arable wheat field (with bacteria-based food webs).

Rewetting after drought considerably enhanced soil NO_3^- availability for all cropping systems, similar to observations in grassland drought experiments (Hofer et al., 2017; Klaus et al., 2020). This overcompensation can be explained as exposure of accumulated microbial and plant necromass and previously protected organic matters during rewetting of soils (Borken and Matzner, 2009). Such a pulse of soil NO_3^-, which we found for all cropping systems with fertilized but also unfertilized crops, can potentially lead to sudden nitrate leaching and harm groundwater safety (Gordon et al., 2008; Khan et al., 2019). Thus, a decelerated release of nutrients from easily degradable plant litter would be beneficial for groundwater safety and would better meet the nutrient demands of the following crop. In this regard, further long-term studies are needed because we only tracked decomposition a few weeks after drought until crop harvest, but not beyond. Potentially, the drought effect especially for low-quality litter could still be compensated before the next crop is established. However, it is important to note that both standard litters used in this study originated from woody species, and therefore are of lower quality than most tissue of arable crops, which frequently include legumes. Such differences in crop species affect not only the quality but also the quantity of litter brought into the soil, further affecting litter decomposition at the field scale.

4.4. No general drought legacy effect on decomposition

Some days after the drought was relieved the soil was rewetted by ambient precipitation and we did not observe a general *drought legacy effect* for litter introduced right after shelter removal (Fig. 5b). Yet, this was variable among the three crops studied, as in pea-barley and maize, the decomposition of especially green tea was still reduced due to the prior drought. This potential legacy effect will be highly depending how quickly the soil was rewetted after shelter removal, indicating the actual end of the drought.

As there was no general *drought legacy effect*, we conclude that the negative drought effects on the soil microbiome were rather transient and the process of decomposition by itself was quickly recovering. This is supported by previous studies showing that soil microbes became active again within few hours after rewetting (Placella et al., 2012) and recovered within one to several days (Meisner et al., 2015, 2013). Whether the assumption that the soil microbial system recovers soon after rewetting without further functional changes holds true needs further research, jointly analyzing drought effects on litter decomposition and soil microbial communities at high temporal resolution. Despite drought effects on litter decomposition in short-term being transient, drought could have critical impacts on the synchronizing of N supply and plant N demand (Ullah et al., 2019).

5. Conclusion

Litter decomposition was strongly affected by drought and litter quality, but not by the four cropping systems studied, including organic farming and the (indirect) effects of conservation tillage. Considerably reduced litter decomposition due to simulated drought demonstrates that drought had a major impact on soil functioning. Thus, although we found no general drought legacy effect, our study suggests litter decomposition to be a highly sensitive process when it comes to an increasing severity of drought events in the future, regardless of farmers' short-term choice for a specific cropping system (time of system change <10 years). As we further found the decomposition of high- compared to low-quality litter to be less resistant to drought, but more resilient after drought, the decomposition of high-quality litter may approach pre-disturbed conditions earlier than that of low-quality litter. Thus, management options that improve litter quality, such as the use of legumes as main or cover crops, could be an option to enhance the resilience of litter decomposition against drought.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.agee.2022.108078.

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