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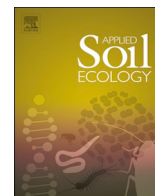
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## Soil compaction effects on litter decomposition in an arable field: Implications for management of crop residues and headlands

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## ABSTRACT

Soil compaction is a major threat to agricultural soils. Heavy machinery is responsible for damaging soil chemical, physical and biological properties. Among these, organic matter decomposition, which is predominantly mediated by the soil biota, is a necessary process since it underpins nutrient cycling and the provision of plant nutrients. Understanding factors which impact the functionality of the biota is therefore necessary to improve agricultural practices. To better understand the effects of compaction on the soil system, we determined the effects of soil bulk density and soil penetration resistance on the decomposition rates of litter in three distinct field zones: a grass margin, sown at the edge of the field adjacent to the crop, tramlines in the crop:margin interface, and crop. Three litters of different quality (ryegrass, straw residues and mixed litter) were buried for 1, 2, 4 and 6 months in litter bags comprising two different mesh sizes (0.02 and 2 mm). Bulk density and soil penetration resistance were greater in the compacted tramline than in the margin or the crop. The greatest amount of litter remaining in the bags after 6 months was found in the tramline, and the least in the grass margin. Differences between treatments increased with burial time. No significant differences in mass loss between the two mesh sizes was detected before the fourth month, implying that microbial activities were the main processes involved in the early stages of decomposition. Decomposition in the tramline was clearly affected by the degradation of soil structure due to heavy compaction. This study shows that soil conditions at the edges of arable fields affect major soil processes such as decomposition. It also reveals the potential to mitigate soil degradation by managing the headland, the crop residues and the machinery traffic in the field.

## 1. Introduction

Land-use is a primary determinant in driving soil processes (Holland et al., 2014; Postma-Blaauw et al., 2010; Sousa et al., 2004). It has been shown that vegetation cover modifies soil biodiversity (Crotty et al., 2015, 2014) and that the more intense the land use (intensive crop production in comparison to extensive grassland), the fewer the number of functional groups prevail (Tsiafouli et al., 2015). In 1994, the United Kingdom government published a Biodiversity Action Plan, establishing arable field margins at the edge of fields as priority habitat (Maddock, 2008) and supported by a new environmental stewardship scheme for farmers to increase and support biodiversity in the agricultural landscape in 2014 (Department for Environment, Food and Rural Affairs,

2014). This included compensation for the setting up of grass margins around arable fields with the primary aim of encouraging aboveground biodiversity (Department for Environment, Food and Rural Affairs, 2014; Meek et al., 2002). Evidence suggests such margins can provide important ecosystem services including pollination and pest management (e.g. Lu et al., 2014). However, the implications for the below-ground biodiversity and the functions they support have been considered less, even though it has been shown that the soil biota can be adversely affected by field management (Sechi et al., 2017). Field margins affect nutrient transformation and run-off (Marshall and Moonen, 2002), and the soil fauna plays a pivotal role in many of the soil processes that, in turn, deliver ecosystem services (Bardgett and van der Putten, 2014; Wall et al., 2015). Among these services,

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decomposition, a biologically-driven process, underpins nutrient cycling and primary production (Coleman et al., 2004; Hättenschwiler et al., 2005). The interaction between different classes of organisms (microbiome and macrobiome) is necessary to ensure the decomposition of primary organic matter (Bradford et al., 2002). Although the role of the microbiome (bacteria, archaea and fungi) is reasonably well understood, Setälä et al. (1996) demonstrated the benefits of a more complex community for improved nutrient cycling. It has also been shown that macrofauna modify the processes of decomposition by their action on the microbiota (Hättenschwiler et al., 2005; Joly et al., 2015). Due to its impact on organic matter dynamics (Wolters, 2000), the presence of the macrobiome (meso- and macrofauna) is required to achieve the decomposition of plant litter and therefore should be regarded as a potential tool for crop management and nutrient cycling in agricultural contexts. In such contexts, most of the litter decomposition is affected by agronomic practices, such as management of crop residues, fertilisation or soil compaction from agricultural machinery. The amount and quality of organic matter returned to the system (Fierer et al., 2005; Gergócs and Hufnagel, 2016; Milcu and Manning, 2011) together with the presence of faunal and microbial communities (Murray et al., 2009) are primary factors regulating decomposition rates. Thiele-Bruhn et al. (2012) noted the capability of agricultural practice to control the quality of primary organic matter entering soil systems and therefore its capacity to modify the soil community and its activity. To understand the effects of litter quality, Johnson et al. (2007) tested the decomposition of five crops of varying chemical composition and three different organs of each plant, and showed that crop and plant parts affected decomposition rates and C-pools at the soil surface. This implies some potential for agricultural soil management via crop residues.

The architecture of the habitat and the associated propensity for belowground oxygen supply (modulated by the soil pore networks) are two more factors affecting decomposition rates. The deterioration of soil structure (principally via a reduction in porosity and connectivity of pores) caused by external factors has been shown to affect microbial mineralisation (Beylich et al., 2010; De Neve and Hofman, 2000), as well as habitat and food resources that support the soil fauna (Beylich et al., 2010; Althoff et al., 2009; Larsen et al., 2004). In agricultural landscapes, soil structure is exposed to deterioration by heavy machinery traffic and many arable soils are sensitive to increased compaction, causing a decline in crop yield (Hamza and Anderson, 2005). Within the scope of environmental schemes, and to prevent damage to improved biodiversity habitats such as field margins, the policy requires that farmers do not manoeuvre on the field margins, obliging them to turn at the edges of the crop and thus creating a compacted area between the main crop and the margin. A better understanding of the effects of compaction on organic matter decomposition and biological activity in soils is a necessary step to improve soil management in

agricultural systems and to mitigate the impacts of compaction.

In this study, we determined organic matter decomposition rates of plant material (wheat straw and ryegrass residues) in contrasting zones of an arable field that had been subjected to different pressures. We aimed to identify effects of machinery wheeling and agricultural management on decomposition and understand how the response changes with respect to litter type and soil faunal exclusion. We hypothesised: (i) decomposition rate would be lowest in more compacted soils; (ii) ryegrass litter, because of its lower C:N ratio, would decompose faster than straw residues, the decomposition rate of an equal mixture of both litters would fall within the interval between the two; (iii) exclusion of the soil meso- and macrofauna would reduce the decomposition rate.

## 2. Materials and methods

### 2.1. Site and soil characteristics

The experiment was carried out between October 2016 and April 2017 at The Grange Farm, Northamptonshire, United Kingdom (52° 18' 2.73" N; 0° 45' 52.83" W) in an arable field planted with oilseed rape (*Brassica napus* L.) which had previously been in winter wheat (*Triticum aestivum* L.). The field had been managed using minimum tillage techniques (i.e. no deep ploughing) for at least 15 years. Mineral fertilisation and chemical inputs were applied to the crop following the UK standard scheme management for farmers (Agriculture and Horticulture Development Board, 2017). The crop was planted in a field bordered by a 6 m-wide 10-year-old grass margin that had been set up to promote biodiversity in the agricultural landscape (Department for Environment, Food and Rural Affairs, 2014; Maddock, 2008). The soil was classified as Hanslope series, a typical calcareous pelosol from a clayey chalky drift series (calcaric stagnic cambisol soil) with poor drainage capacity and high sensitivity to compaction (Cranfield University, 2017).

During the period of study climate conditions were characterised by two dry periods; one in October at the beginning of the experiment and one in December. Temperatures were normal for the region (Fig. S1).

The experimental area consisted of 18 plots (6 × 6 m) distributed among six blocks along the south side of the field. Each block comprised three plots, one in each of three field 'zones': one in the grass margin, one in the tramlines between the margin and the crop, which were visibly compacted, and one in the actual crop. The field zones (margin, margin-crop interface and crop) were spatially constrained and so randomisation of plots within blocks was not possible. Total soil carbon (C) and total soil nitrogen (N) concentrations were measured using an elemental analyser (N1500, Carlo Erba, Milan, Italy). C:N ratio was determined as average values calculated from cores taken at each of the 18 plots.

**Table 1**

Average values (n = 6) and standard error (*± SE* in italic) of various soil properties measured in three zones of an oilseed rape field (October 2016). Superscript letters <sup>a</sup> and <sup>b</sup> show significant difference of means between the field zones (Least square difference test, Bonferroni adjustment).

Field zones		Water content (% Volume of Soil <sup>-1</sup> )	Bulk density (g.cm <sup>-3</sup> )	Total C (% Volume of Soil <sup>-1</sup> )	Total N (% Volume of Soil <sup>-1</sup> )	C:N ratio
Grass margin	Mean	17.4 <sup>a</sup>	0.89 <sup>a</sup>	4.08 <sup>a</sup>	0.40 <sup>a</sup>	10.20 <sup>a</sup>
	<i>± SE</i>	1.02	0.03	0.39	0.02	0.39
Tramline	Mean	11.7 <sup>b</sup>	1.25 <sup>b</sup>	2.32 <sup>b</sup>	0.27 <sup>b</sup>	8.61 <sup>b</sup>
	<i>± SE</i>	0.62	0.03	0.14	0.01	0.46
Crop	Mean	14.6 <sup>a</sup>	1.02 <sup>a</sup>	2.36 <sup>b</sup>	0.25 <sup>b</sup>	9.58 <sup>b</sup>
	<i>± SE</i>	0.25	0.06	0.19	0.02	0.42

## 2.2. Soil compaction assessment

Soil bulk density (Laryea et al. (1997) was determined from cores (8 cm diameter × 10 cm depth) taken at random from each of the 18 plots at the beginning of the experiment. This sampling method was considered appropriate for our requirements as it has been shown to not significantly affect bulk density measurement (Özgöz et al., 2006; Page-Dumroese et al., 1999). Samples were dried at 105 °C for 24 h, then plant residues and stones were removed by 2 mm sieving. Water content was calculated from the proportion of dry soil to wet soil (Table 1). Soil penetration resistance was recorded on April 1st 2017 with a penetrometer (Solutions for Research Ltd, Silsoe, Bedfordshire, UK) fitted with a 9.45 mm diameter (base area  $7 \times 10^{-5} \text{ m}^2$ ), 30-degree cone. At every sampling point, the soil penetration resistance was measured at 14 depth points each 3.7 cm apart (from 3.7 cm to 51.8 cm depth). Penetrometer resistance was calculated by dividing force at each depth by the cone base area. Ten replicate measurements were randomly taken at each plot. Data were calibrated by Solutions for Research Ltd, Silsoe, Bedfordshire, UK; and soil strength (Pa) was determined from the force measured by the penetrometer ( $\text{kg} \times 9.81$ ), divided by the base area ( $\text{m}^2$ ), so that a 10 kg force reading represented 1399 KPa.

## 2.3. Organic matter decomposition experiment

Litter bags (6 cm length × 5 cm height) were made using two mesh sizes; one set with a plastic mesh size of 2 mm allowed full access by the soil biota, and one set with 0.02 mm nylon mesh which excluded most of the fauna and allowed microbial access only.

Three types of litter of different quality (C:N ratio) were prepared: a low C:N ratio perennial ryegrass (*Lolium perenne* L.), a high C:N ratio wheat straw (*T. aestivum*) and a 50:50 mixture of both types of litter of intermediate C:N ratio. Ryegrass and wheat straw were oven dried to constant weight at 105 °C. Straw and ryegrass were chopped into approximately 1.5–2 cm length pieces, then, 1.0 g of the litter was added to each of the litter bags (0.5 g of both chopped litter types was added for the mixed litter treatment). Litter bags were carefully stored in individual boxes to prevent physical damage during transportation; litter loss (collected in the bottom of boxes after transportation) was considered insignificant in terms of affecting future measurement. Average values of total carbon and total nitrogen of wheat straw and ryegrass were measured from 5 subsamples of each of the initial materials (i.e. T0), and after 6 months (T6) from material remaining in both the small and large mesh bags, for each of the three litter types (ryegrass, straw and a 50:50 mixture of both) using an elemental analyser (N1500, Carlo Erba, Milan, Italy). The initial C:N ratio of the 50:50 mixture was taken as the arithmetic mean of the constituent ratios.

In total, 432 litter bags were prepared, half with the small mesh size and half with the large mesh. We allocated 24 litter bags to each of the 18 plots (four bags of each mesh size containing straw, rye or the 50:50 mixture) and buried these on 1 October 2016. A sub-set of 108 bags (one bag of each treatment combination from every plot) were removed on 1 November 2016, 1 December 2016, 1 February 2017 and 1 April 2017, representing 1, 2, 4 and 6 months' burial duration. The latter time is consistent with the cropping cycle, when the cultivated part of the field is then physically disrupted. Litter bags were buried in the top soil at 5 cm depth in each plot and the position of bags of each treatment combination was completely randomised within the plot. To minimise disturbance and to preserve the context of the inherent soil structure as far as possible, a vertical slit was made with a spade, just sufficient to locate the bag, and then closed up by firming the soil back into position. A string and a knot code system were used to identify each treatment. One bag was missing on the first and third collection dates, and 5 bags were missing on the last date.

After removing the litter bags from the ground, the litter was removed from the bags; soil particles were gently washed away from the litter using a 15 µm sieve to retain plant materials. The litter was then

dried and weighed as described above. The proportion of litter remaining following each burial duration was then calculated.

## 2.4. Statistical analyses

Impacts of field zone on bulk density was estimated by a 1-way analysis of variance (ANOVA). An analysis of covariance (ANCOVA) was used to test the effect of field zone (grass margin, tramline and crop) on soil penetration resistance, controlling for the effects of depth, which co-vary with the field zone effect. We tested the similarity of soil penetration resistance in the “grass margin” and the “crop” zone by using a nested treatment structure in the model; the ANCOVA therefore the effect of the uncompacted zones (grass margin and crop) versus the compacted tramline as well as comparing the effect of grass margin versus crop.

The five samples taken from the initial litter material (T<sub>0</sub>) are pseudoreplicates in terms of testing for treatment effects and as such were not analysed statistically. However, means and standard errors have been presented as a basis for comparison between litter types and for comparing later time points to baseline. A two-way split plot analysis of variance (ANOVA) was used to test for the differences of C:N ratio between two treatment factors (mesh size and litter type) and their interaction. We used a post-hoc comparison Tukey test at 95% confidence level to see where differences between factors occurred.

A four-way split plot ANOVA was used to determine effect of the treatment factors (mesh size, litter type, field zone and time period in the ground) and their interactions on the quantity of litter remaining at the end of the experiment. Because of the destructive sampling of the litter bags, time was not considered as a repeated measurement. We used Fisher's Least significant difference method (LSD) to assess whether pairwise combinations were different from one another, with Bonferroni adjustment. Similarly, the effect of mesh size, litter type and field zone were analysed using a 3-way ANOVA for month 1, 2, 4 and 6 separately. All statistical analyses were done using R software 3.1.2 (<http://www.r-project.org/>).

## 3. Results

### 3.1. Soil compaction

Bulk density was significantly greater in the compacted area of the tramline compared to either the grass margin or the crop (1-way

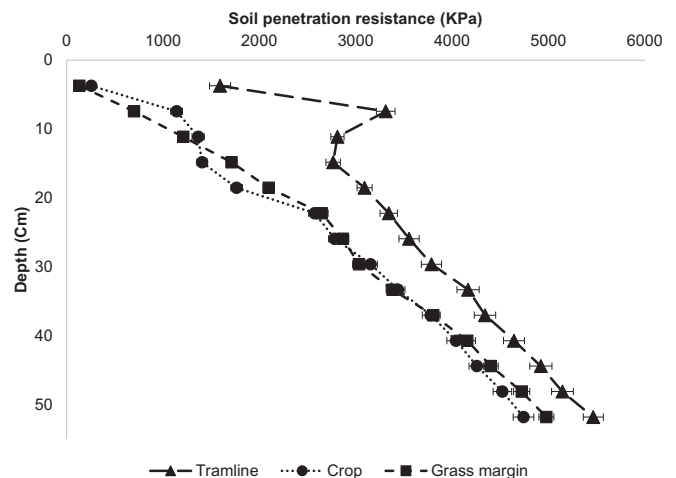
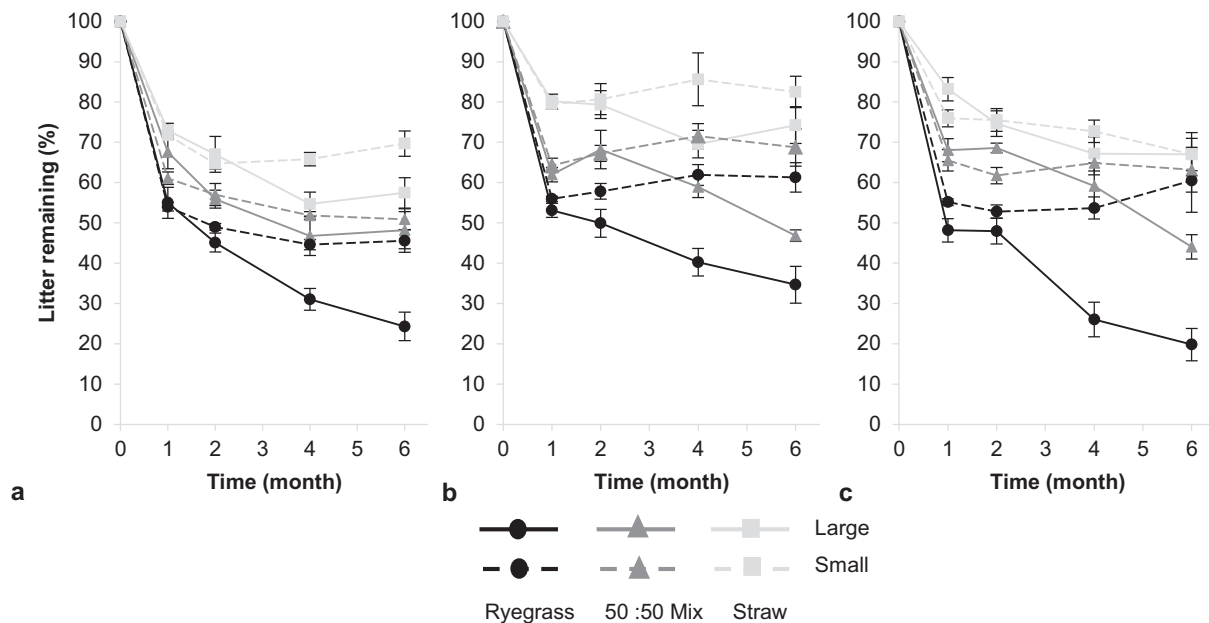


Fig. 1. Profiles of soil penetration resistance (KPa) within three field zones (grass margin ■, tramline wheeling in the crop-margin interface ● and, crop ▲) at 14 depth points (3.7–51.8 cm depth) within a field containing oilseed rape, 2017 cropping season. Points show means ( $n = 60$ ); bars denote standard error. In some instances, these fall within the confines of the symbols.



**Fig. 2.** Percentage of different litter types (perennial ryegrass (*Lolium perenne*) black ●, wheat straw (*Triticum aestivum*) light grey ■, and a 50:50% mixture of both litters grey ▲) remaining after 1, 2, 4 and 6 months buried in three different zones of a field containing oilseed rape, in litter bags with small (0.02 mm) and large (2 mm) mesh sizes; Zones: grass margin (2a), compacted tramline (2b) and crop (2c). Month 0 corresponds to the start of the experiment (1st October 2016) and Month 6 the end of the experiment (1st of April 2017). Points show means (n = 18); bars denote standard error. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

ANOVA  $F_{(2,13)} = 18.6$ ,  $P < 0.001$ ; Table 1).

The soil penetration resistance increased significantly with depth in all of the three field zones (ANCOVA  $F_{(1,246)} = 1003$ ,  $P < 0.001$ ). A peak was observed at 7.4 cm in the tramline, whereas the slope of the resistance in the crop increased below the historical ploughed layer at 23 cm depth (Fig. 1). There was no difference in soil penetration resistance between the crop and the field margin zones (ANCOVA  $F_{(1,246)} = 0.23$ ,  $P = 0.63$ ) and overall, the soil penetration resistance was significantly greater in the compacted zone (tramline in the crop-margin interface) than in the uncompacted zones (crop and field margin zones combined) (ANCOVA  $F_{(1,246)} = 129$ ,  $P < 0.001$ ).

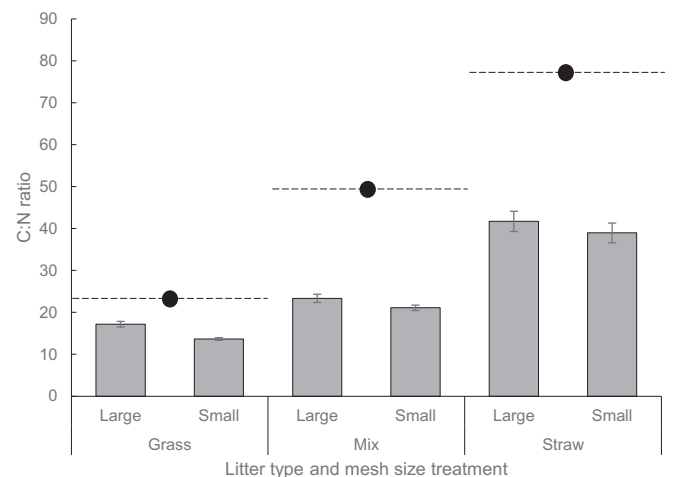
### 3.2. Litter decomposition

#### 3.2.1. Comparison of the two mesh sizes

In the first two months of the experiment, regardless of the field zone or the litter type, there was no significant difference in litter remaining between the large and small mesh size bags (3-way ANOVA for Month 1 and Month 2,  $F_{(1,74)} = 0.63$ ,  $P = 0.431$  and  $F_{(1,75)} = 0.67$ ,  $P = 0.415$ , respectively). However, from Month 4, there was generally more litter remaining undecomposed in the small mesh than in the large mesh size bags (3-way ANOVA  $F_{1,74} = 69.3$ ,  $P < 0.001$ ; Fig. 2). This effect was persistent at Month 6 (3-way ANOVA  $F_{(1,70)} = 92.7$ ,  $P < 0.001$ ). Overall the combined effect of mesh size on litter remaining over time was significant (4-way ANOVA  $F_{(3,350)} = 34.84$ ,  $P < 0.001$ ). The effect of the field zone combined with the mesh size was also significant (4-way ANOVA  $F_{(2,350)} = 3.65$ ,  $P = 0.027$ ), with relatively less litter lost in the large compared to the small mesh bags when these were buried in the tramline or the crop rather than in the grass margin zone (Fig. 2).

#### 3.2.2. Effects of crop litter quality

Litter type significantly affected the proportion of plant material remaining in the bags at the end of the experiment (4-way ANOVA  $F_{(2,350)} = 386$ ,  $P < 0.001$ ); A mean across treatments  $\pm$  standard error (SE) of  $72.6 \pm 7.3\%$  of the straw remained after 6 months, while



**Fig. 3.** Carbon:nitrogen ratios of the three plant residues used in litter bags of large (2 mm) and small mesh size (0.02 mm) at the outset of the study (T0), and main effects of litter type and mesh size after 6 months burial (T6). The grey bar charts display C:N values at T6 (error bar showed standard error), while the black dots and the broken lines denote C:N ratios at T0. For the mixture, this is calculated as arithmetic mean of ryegrass and straw values based upon a 50:50% mix of these constituents. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

$47.1 \pm 8.1\%$  of the ryegrass was left. Mixed litter had an intermediate decomposition rate, with  $60.4 \pm 8.0\%$  of material remaining. There was a significant interaction of the combined effects of litter type and mesh size (4-way ANOVA  $F_{(2,350)} = 22.4$ ,  $P < 0.001$ ), with less ryegrass litter remaining in the large litter bags than straw residues or mixed litter in small and large mesh size bags (Fig. 2). Even though the difference in litter remaining between the two mesh sizes at Months 1 and 2 was not significant, remaining ryegrass litter in the large mesh size litter bags was marginally smaller than in other treatments (3-way ANOVA  $F_{(2,74)} = 3.08$ ,  $P = 0.052$  and  $F_{(1,75)} = 2.94$ ,  $P = 0.059$  for



Month 1 and Month 2, respectively; Fig. 2).

The C:N ratio of the litter declined over time only in the case of straw and the straw:ryegrass mixture (Fig. 3). After 6 months burial duration, there was no significant interaction between litter type and mesh size, but a highly significant main effect due to litter type (2-way ANOVA  $F_{(2,80)} = 156$ ,  $P < 0.001$ ), where the proportion of difference between  $T_0$  and  $T_6$  was around three-fold greater in the case of straw compared to ryegrass (Fig. 3). In the case of the mixture, the C:N ratio was about half that of the straw, but significantly greater than that for ryegrass alone (Fig. 3). There was also a significant effect of mesh size upon C:N ratio after 6 months where the ratio was 10% smaller in small compared to large mesh bags (2-way ANOVA  $F_{(1,75)} = 5.30$ ,  $P = 0.02$ ; Fig. 3).

### 3.2.3. Effect of the field zone on litter losses

The location of the litter bags in the field (zone) significantly affected the decomposition rate of all litter types within bags of the two different mesh sizes (4-way ANOVA  $F_{(2,10)} = 34.0$ ,  $P < 0.001$ ). With a mean of  $64.8 \pm SE 7.8\%$ , remaining litter was greater in bags placed in the tramline and similar decomposition rates were observed in bags buried in the grass margin and in the crop (on average 55.0% and 60.2% of litter remaining, respectively).

## 4. Discussion

We hypothesised that deteriorated soil conditions, caused by trafficking, would reduce plant litter decomposition at the interface between the crop and the margin in comparison to the grass margin (Hamza and Anderson, 2005). We used soil bulk density as a simple surrogate to indicate the pore space, soil compaction (Buckman and Brady, 1960) and therefore inappropriateness of habitat and conditions for soil life (Beylich et al., 2010). The greatest values of bulk density were observed in the tramlines, where decomposition was the slowest. Horn et al. (1995) observed that greater bulk density caused a decrease in soil aeration, which in the case of the margin:crop area of this study might indicate a slow decomposition rate and explain the observation of more material remaining in the bags at the end of the experiment. Although it was not possible to disentangle physical losses of litter from chemical decomposition, we argue that the whole process of decomposition includes chemical processes as well as physical movement of the litter (which can be caused by physical changes in soil, such as compaction, or biological transposition e.g. by earthworms). In the case of the tramline, we concluded that the limited losses of litter were a consequence of compaction; whether it resulted from physical constraints (restricted pore space resulting in limited losses of material or fauna movements), or biochemical constraints (limited oxygen supply to allow efficient microbial decomposition) should be established via controlled experiments. Compaction over the whole soil profile was assessed by taking soil penetration resistance measurements. Penetrometer data is a measure of soil strength (Bengough et al., 2000), here implying that the compacted soil was stronger than that associated with crop and grass, which were not different from each other on the day the measurements were made. Although only soil strength at the soil surface was used as a data to relate to effects of compaction on litter decomposition; all three treatments show an increase in strength with depth which is usual and probably due to the soil overburden (Horn et al., 2007). The increased strength may be due in part to differences in water content and or bulk density (Bengough et al., 2000), but whatever the underlying causes of soil strength, the compacted soil in the tramlines was stronger than in the grass margin or in the crop (Fig. 1). All the penetrometer curves were the same basic shape apart from the deviation at 7.4 cm depth in the tramline, which could be explained by wheeling pressures compacting the soil (Fig. 1). This could be interpreted as an indicator of degraded soil condition (Duiker, 2002) and results in an impermeable layer of soil preventing water drainage, increasing the likelihood that water capacity over the winter season

would be exceeded and consequently the limited supply of oxygen would restrict decomposition processes (Beylich et al., 2010; Horn et al., 1995; Whalley et al., 1995). The shallow angle of the slope observed in the soil penetration resistance measurements from the cropped zone corresponds to the historical ploughed layer at 23 cm. Above this layer, soil penetration resistance in the grass margin and the crop zone behaved differently but reached similar intensities below this interface. Even though the field had been farmed using minimum tillage cultivation techniques for the past 15 years, this could reflect the long-term effect of previous ploughing practices on soil structure. The potential impact of this on soil processes (e.g. Peigné et al., 2013) warrant further investigation. Our results showed that decomposition occurs more slowly in the compacted soil of the tramlines at the crop-margin interface regardless of the litter type or the mesh size of the bags used in the experiment.

The two different mesh sizes of litter bags used in the decomposition experiment enabled conclusions to be drawn about the effects of microbial communities and larger soil fauna on decomposition since the large mesh size allowed access of all soil fauna and the small mesh size excluded this fraction and would therefore be predominantly microbial (Bokhorst and Wardle, 2013). Before Month 4, there was no difference in mass loss between litter bags of the two mesh sizes, implying that initial decomposition (Month 1 and Month 2) was primarily carried out by microbes or that effect of the soil fauna was negligible. In this study, the addition of an exogenous source of organic matter might have stimulated primary microbial colonisation, resulting in mineralisation of the fresh organic matter, leaving humified organic matter (Wardle and Lavelle, 1997). Over time, the mass loss of litter in the large mesh size bags was greater than in the small mesh size bags, implying that the activity of larger invertebrates become significant as they break down this recalcitrant pool of organic matter, making it available to mineralisation (Bradford et al., 2002; Schädler and Brandl, 2005). Carrillo et al. (2011) observed that changes in litter decomposition processes, induced by the presence of meso- and macro-invertebrates, was time dependent and highlighted the importance of temporal dynamics in effects on the soil fauna. The amplitude of the difference between decomposition rate in the large and small mesh sizes was the greatest in the compacted tramline, implying that the inclusion of larger soil organisms may have supported litter decomposition despite restricted soil conditions. However, the large mesh size bags buried in the grass margin showed contrasted results and no difference in litter loss was observed compared to the small mesh size. This suggests that the presence of ryegrass in the mixed litter, in an environment where microbial communities are conditioned to decompose grassy residues, would have a synergistic effect on the decomposition of the straw from the mixture. This is a phenomenon that we could attribute to a 'priming effect' from the ryegrass litter to amplify the decomposition of the straw in the mixture (Fontaine et al., 2003). In the small mesh size bags, such effect would therefore compensate for the effect of the soil fauna inclusion observed in the large mesh size bags. Unlike the mixed litter, straw residues alone decomposed faster in the large than small mesh size bags. This contrasting effect could result from the inability of the microbial populations in the grass margin to instigate decomposition of wheat straw without a priming effect.

Litter quality (expressed here as C:N ratio) is well established as a driver of decomposition (Hamza and Anderson, 2005; Wardle and Lavelle, 1997) and accordingly in this study, the decomposition rate was influenced by litter type and its quality; the greater the C:N ratio of the litter, the slower the decomposition. After 6 months in the soil, significantly more litter remained in the bags containing wheat straw than those containing ryegrass. Decomposition of mixed litter occurred within the interval between the wheat and ryegrass treatments; although the decomposition rate of the mixture was affected differently by synergistic, antagonistic or additive effects of the residues depending on the field area or the mesh size. It was shown by Redin et al. (2014), that the diversity of functional and chemical traits of mixtures of crop

residues (regarding the plants alone) influences decomposition rates of residue mixtures. Because the effect of the mixed litter on decomposition rates was null only in the small mesh size treatment - where only microbial decomposition occurred - it might be evidence for the 'resource concentration hypothesis' presented by Pan et al. (2015). This posits that the diversity of plants in a litter mixture decelerates decomposition of litter because decomposers of each species suffer from a reduced availability of their preferred food resource. Because this was not observed in the large mesh size litter bags, it implies the role of larger soil invertebrates regulating and promoting the microbial decomposition (García-Palacios et al., 2013; Schädler and Brandl, 2005). After 6 months decomposition, we observed that the C:N ratio of the straw and the mixture diminished while the C:N ratio of the ryegrass did not change. We posit that lower C:N ratio materials, such as ryegrass, would decompose faster than high C:N ratio materials, such as straw; however, we showed that changes in C:N ratio over time were independent of decomposition rates and that the amplitude of C:N ratio changes between T0 and T6 depended on the initial C:N content. This change may have been driven by proximity of the litter to the soil organic matter (SOM) C:N value; the closer to the SOM C:N value, the smaller was the amplitude of change of the litter C:N value. For example, the initially high C:N value of straw had a greater amplitude of change than the C:N value of the ryegrass, and tended towards an equilibrium, similar to the average C:N ratio 11.5 found in cambisol soils (Batjes, 1996). Contrary to the decomposition rate, there was a greater C:N ratio at the end of the experiment in the large mesh size bags than in the small ones, evidence that most of the chemical decomposition was effected by the microbial community, while the physical breaking down of the litter was as a result of the action of larger organisms (Bradford et al., 2002).

We have shown that prevailing soil conditions at the edge of arable fields affect major soil processes such as decomposition. Soil porosity is particularly affected in this area due to heavy machinery traffic, and inputs (fertilizers, crop residues) are less homogeneously distributed here than in the middle of the field. The uneven management and the increased disturbance at the edge of the field are probably causal factors of the observed lower crop yields in this area. For instance, Sparkes et al. (1998) recorded 3–19% less yield at the edge than in the middle of cereal fields and Wilcox et al. (2000) reported high variability in yield in the same zone of winter wheat fields. This results in a "sensitive zone" between the margin and the crop where soil biological and chemical dynamics are reduced if not appropriately managed. However, this study also revealed that there is potential to mitigate the effects of compaction in this sensitive zone. We have shown that the quality of organic amendments (low C:N ratio) can partially mitigate the lower decomposition rates in the compacted zone, yet the inclusion of larger soil organisms helped to support litter decomposition in this specific zone. As shown by Baveye et al. (2016), both the characteristics of the habitat and the structure of the soil fauna community living there are of importance in sustaining soil ecosystems. The presence of low C:N ratio crop residues, such as grass cover or cover crops, would sustain an adequate community of soil invertebrates that could facilitate decomposition of main crop residues and support soil structure in the field area subjected to compaction.

We underline the important role of soil-dwelling invertebrates in the decomposition process. In the current United Kingdom subsidy schemes, farmers are paid to manage crop margins to enhance botanical diversity, thereby supporting farmland birds and pollinators (Department for Environment, Food and Rural Affairs, 2014; Hatt et al., 2017; Kovács-Hostyánszki et al., 2017; Mansion-Vaquíé et al., 2017). These schemes also tend to benefit soil functions supported by the belowground diversity (Frazão et al., 2017; Roarty and Schmidt, 2013; Smith et al., 2009), but the resulting compacted zone, created by machine turning in the tramlines of the margin-crop interface (as operations are not allowed on the margins), impairs the ability of soil to function (Arvidsson and Håkansson, 2014; Beylich et al., 2010; Hamza

and Anderson, 2005). The ban on driving on the field margin exacerbates this. One simple option would be to increase the current width of the margin supported by agri-environmental schemes from 2 to 6 m to 12 m, and allow turning on this additional area. Grasslands are more resistant to compaction (Matthews et al., 2010) and we believe that such a system would minimise the "sensitive zone" and maintain soil processes such as decomposition despite the high pressures and disturbances applied in the cropped area.

This study highlights that the current regulations for the use of grass margins could be modified to optimise the ecosystem services they provide. We propose that adapting the rules regarding grass margins could result in a combined benefit for growers and ecosystem services. For instance, extending the field margin over the compacted tramline and allowing farmers to drive and turn in this extra-margin could result in improvement of soil structure, increase in above and belowground biodiversity, enhancement of ecosystem services, and reduction of the costs resulting from farming this non-profitable part of the field, thereby contributing to achieving more sustainable food production systems.

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### Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.apsoil.2018.10.004>.

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