Grazed temporary grass-clover leys in crop rotations can have a positive impact on soil quality under both conventional and organic agricultural systems

Running title: Crop-livestock & leys improve soil quality

Caio F. Zani[a*], John Gowing[a], Geoffrey D. Abbott[a], James A. Taylor[b], Elisa Lopez-Capel[a], Julia Cooper[a]

[a] School of Natural and Environmental Sciences, Newcastle University, Kings Road, Newcastle upon Tyne, England, NE1 7RU, United Kingdom.

[b] UMR ITAP, University of Montpellier, INRAE, Institut Agro, Montpellier, 34000, France.

* Corresponding author: Caio Fernandes Zani, School of Natural and Environmental Sciences, Newcastle University, Kings Road, Newcastle upon Tyne, England, NE1 7RU, United Kingdom

E-mail address: caiofzani@gmail.com

This article has been accepted for publication and undergone full peer review but has not been through the copyediting, typesetting, pagination and proofreading process which may lead to differences between this version and the Version of Record. Please cite this article as doi: 10.1111/ejss.13002

Abstract

Soil quality (SQ) is the ability of soil to provide ecosystem functions and services. Implementation of a certain agricultural system can affect SQ and therefore play an essential role in achieving sustainable agriculture. The aim of this study was to explore how agricultural systems (conventional *vs.* organic), grazing regime (non-grazed *vs.* grazed) and the different proportions of temporary grass-clover leys in crop rotations (ley time proportion-LTP) affect SQ within a mixed (cropping and pasture/dairy system) commercial farming enterprise in the UK. Seven SQ indicators were evaluated, including chemical (pH; available phosphorus-P; potassium-K), physical (bulk density-BD; aggregate stability-AS) and biological (total carbon-C; microbial biomass carbon-MBC) sectors. All SQ indicators were measured at three depth intervals (0-0.15, 0.15-0.30, 0.30-0.60 m), except for AS and MBC, which were only considered for the topsoil (0-0.15 m). The findings reflected existing knowledge on the advantages of organic *vs.* conventional systems on SQ indicators, with the former showing higher MBC and similar K, BD, AS and C in the 0-0.30 m compared to the latter. Lower topsoil available P in organic systems can be related to the lack of measurements in all P pools. When grazing was included i) both agricultural systems showed higher topsoil available P, C and MBC; and ii) there was a higher topsoil K in organic systems whilst it positively affected topsoil BD and C (0.15-0.30 m) in conventional systems. Increasing LTP to 30-40% of the full crop rotation increased topsoil AS and C (0-0.30 m) in a linear fashion. Subsoil conditions (> 0.30 m) favoured K, BD and C in conventional systems, but these results should be considered carefully. It was concluded that both organic and conventional systems delivered similar levels of SQ and that reviving mixed farming systems may be a key factor for delivering multi-functional agroecosystems that maintain SQ and optimise ecosystem services.

Keywords: ecosystem services, land management, ley-arable rotation, mixed farming, soil functions, soil health.

Highlights

- Single-farm comparison of top- and subsoil quality in organic and non-organic systems.
- The organic system increased microbial biomass carbon but decreased topsoil available phosphorus.
- Grazing increased topsoil available phosphorus, carbon concentration and microbial biomass carbon.
- Temporary leys in rotations increased topsoil aggregate stability and carbon concentration.
- Mixed farming is a key factor for delivering multi-functional agroecosystems.

1. Introduction

While intensification of agricultural activity in the last century has supported rapid growth in the global population, it has also contributed to significant environmental impacts. Soil quality (SQ) and thus sustainable agricultural management of soils has become of global interest due to the soil's critical role in providing ecosystem functions and services (Karlen *et al.*, 1997; Doran, 2002; Bünemann *et al.*, 2018). However, there are uncertainties as to how changes in agricultural systems (e.g. from conventional to organic) and the implementation of mixed farming systems (i.e. arable/livestock), with temporary grass-clover leys in crop rotations, affect the SQ of agroecosystems and consequently the environment.

Discussions on SQ emerged in the 1970s and gained ground when concerns around sustainable agriculture in the mid-1980s attracted public attention. In short, SQ encompasses the capacity of the soil to deliver key functions within a particular ecosystem/land use and to sustain biological productivity whilst maintaining or even improving water and air quality and human, plant and animal health (Karlen *et al.*, 1997; Doran, 2002; Bünemann *et al.*, 2018). Based on this definition, it is impossible to directly measure SQ due to its complexity, but it is possible to pursue SQ to ensure sustainability in any given ecosystem. The SQ status of a given ecosystem takes into account inherent and anthropogenic synergies, with the former related to the process of soil-forming and the latter attributed to land use and agricultural management (Karlen *et al.*, 1997, 2008). Soil indicators are measured soil properties that are sensitive to anthropogenic activities and linked to soil functions and ecosystem services. Therefore, they are normally used to indirectly assess SQ (Andrews *et al.*, 2004). The selection of soil quality indicators is crucial, and they should be sufficiently diverse to represent chemical, physical and biological soil properties; the most studied ones being, total soil carbon (C), pH, phosphorus (P), water storage and bulk density (BD) (Bünemann *et al.*, 2018).

The organic system has been proposed as an attractive agricultural management option to enhance SQ, particularly when compared to non-organic 'conventional' systems (Reganold & Wachter, 2016). Organic systems rely mainly on ecological processes, which strive to support as well as enhance biodiversity and biological cycles, thereby re-establishing ecological harmony (IFOAM, 2012). National organic guidelines include practices that may improve SQ, such as diverse crop rotations, mixed farming systems with high animal welfare standards and genetically diverse animal and plant communities, and limited use of all synthetic input sources. This has been confirmed by studies which have shown positive effects on several soil indicators normally used to assess SQ, such as soil C, soil structure and soil microbial biomass (Maeder *et al.*, 2002; Gattinger *et al.*, 2012; Lori *et al.*, 2017; Cooper *et al.*, 2018; Loaiza Puerta *et al.*, 2018). Other studies have also indicated that when it comes to environmental aspects, organic systems deliver more benefits than conventional systems (Mondelaers *et al.*, 2009; Tuomisto *et al.*, 2012; Meier *et al.*, 2015; Seufert & Ramankutty, 2017). However, organic systems could potentially negatively affect some aspects of SQ, which has led to critics claiming that organic systems will be incapable of feeding the projected global population (Connor, 2008; Pickett, 2013). One of the main concerns is that essential nutrients, such as P and potassium (K), may become deficient under long-term organic systems due restictions on sources of imported crop nutrients (Möller *et al.*, 2018). On the other hand, conventional systems are recognised as having negative impacts on the environment including contributing to greenhouse gas (GHG) emissions (Reay *et al.*, 2012; Stavi & Lal, 2012), decreasing biodiversity (Gomiero *et al.*, 2011; Tsiafouli *et al.*, 2015), increasing pollution of land and water bodies and degrading soil C (Lal, 2004, 2007; Godfray *et al.*, 2010; Amundson *et al.*, 2015), all of which can be linked to declines in SQ.

It has been recognised that no single approach will solve the challenge of achieving future food security (Reganold & Wachter, 2016). Rather, it may be necessary to adopt some farming practices in combination with other strategies. The inclusion of temporary grass-clover leys in crop rotations (a practice usually implemented in organic systems but also currently encouraged under conventional systems) could help to enhanced SQ by regulating the quality and quantity of soil organic matter (SOM) entering the soil system (Paustian *et al.*, 1997). The use of temporary grass-clover leys in crop rotations has also been suggested to improve soil biodiversity, soil C accumulation and nutrient cycling among many other benefits (Lori *et al.*, 2017; Johnston *et al.*, 2017). Recent research has further stressed that if temporary grass-clover leys are grazed (i.e. if the farm is under a mixed arable/livestock system), then there may be an additional benefit to soil C accumulation and enhanced nutrient cycling and utilisation, and consequently improved SQ in the agroecosystem (Chen *et al.*, 2015; Assmann *et al.*, 2017).

Despite the potential benefits of mixed farming systems (arable/livestock), there are still uncertainties regarding two key points: (1) the impact of interactive effects between different agricultural systems (conventional *vs.* organic) and specific practices (e.g. grazing regime: non-grazed *vs.* grazed) on SQ indicators and; (2) the effect of the length of temporary grass-clover leys (in this study referred to as ley time proportion-LTP) in crop rotations on SQ. To address this current gap in knowledge, this study used a mixed commercial farm (cropping and pasture/dairy system), where conventional and organic agricultural systems co-exist, to evaluate the impacts of agricultural systems, grazing regimes and LTP on SQ. The overarching aims of this study were (1) to evaluate the effects of agricultural systems (conventional *vs.* organic), grazing regimes (non-grazed *vs.* grazed) and their interaction on individual SQ indicators and, (2) to assess the effects of LTP in rotations on SQ indicators. The null hypotheses are ultimately that the adoption of the organic system, grazed regime and increases in the LTP do not lead to improvements in any SQ indicators.

2. Materials and Methods

2.1 Study fields selection and description

The study was performed at Nafferton Farm, a mixed (cropping and pasture/dairy system) commercial farm located in North-East England (54º 59' 09'' N; 1º 43' 56'' W, 60 m a.s.l) where both conventional and organic agricultural systems co-exist in a split farm comparison. According to the Köppen classification, the site experiences a marine west coast climatic condition. From 1981 to 2018, the average annual temperature and total precipitation were 8.6 ºC and 638.6 mm respectively (Fig. S1), with a maximum monthly temperature of 22 ºC and a minimum of 0 ºC. The soil is classified predominantly as a Dystric Stagnosol (WRB, 2015); slowly permeable, seasonally wet, acidic loamy to clayey soil that is naturally low in fertility (Farewell *et al.*, 2011; Cranfield University, 2020). Particle-size distribution analysis indicated that the soil samples used in this study had an average of 14%, 45% and 41% of clay, silt, and sand, respectively (sandy silt loam) in the top 0.30 m soil layer, and 21%, 41% and 38% of clay, silt, and sand, respectively (clay loam) in the 0.30-0.60 m soil layer.

Historically, Nafferton farm was a conventional mixed commercial system, with the main activities being a dairy herd, with associated pastoral production, intermixed with a conventional arable cropping system. In 2001, there was a management change from conventional to an organic system across approximately 50% of the farm area (~ 160 ha), while maintaining the mixed (dairy and arable) production system on both the conventional and organic parts of the farm. For the past 14 years, the farm has been run with a mixed conventional and a mixed organic agricultural system side-by-side. Conventional enterprises are operated to current UK best practices (Red Tractor Assurance, 2015) and the organic enterprises to Soil Association (2019) standards. As conventional was the default system for the preceding 50+ years at Nafferton farm, the comparison between the two agricultural systems (conventional and organic) was made using conventional as the baseline. The study fields were deemed suitable since they had similar soil types and experienced similar climatic conditions.

Twelve commercial-sized representative agricultural fields (~ 120 ha of the total 320 ha of the farm) were selected for this study (Fig. 1). Criteria used when selecting the study fields were recent (2008-2017) agricultural system (S) (conventional-CONV *vs.* organic-ORG), grazing regime (G) (non-grazed-NG *vs.* grazed-GG), and crop rotations, i.e. the inclusion of temporary grass-clover leys in crop rotations. In general, agricultural systems (conventional *vs.* organic) were tested using all the twelve study fields, six under conventional and six under organic, which were considered as replicates for each agricultural system. Grazing regime (non-grazed *vs.* grazed), was tested using four non-grazed and eight grazed study fields (two non-grazed and four grazed study fields within each agricultural system, respectively). The stocking rate on the farm is 1-1.5 livestock units ha^{-1}, which was considered to be light to moderate (Soil Association, 2019). Rotations for the organic and conventional agricultural systems did differ slightly, mainly due to the need to have a nitrogen-fixing component within the organic system to support arable production. In addition, ley rotations tended to be longer within the organic system to assist with weed and disease control. As such, it was not possible to have directly paired fields with the same rotational history under the conventional and organic system. Therefore, study fields were deliberately chosen based on the percentage (0 to 100%) of time as temporary grass-clover leys (ley time proportion; LTP), during the previous

10 years and selected within each agricultural system to have a similar spread of LTP (Table 1). In general, the main arable crops grown in the conventional rotation were winter cereals, including winter wheat (*Triticum aestivum*), winter barley (*Hordeum vulgare*) and oilseed rape (*Brassica napus*). Organic rotations included mainly spring wheat and barley and field beans (*Phaseolus vulgaris*). Grass-clover ley periods, in both conventional and organic systems, used a mixture of white and red clover (*Trifolium repens* and *Trifolium pratense*) with perennial ryegrass (*Lolium perenne*). Ley periods in both grazed and non-grazed fields were subjected to two to three harvests for silage per year, depending on their productivity and timing of grazing in the paddock. Further details of management practices in each study field, such as tillage and manure and fertiliser applications, are given in Table 1.

2.2. Soil sampling methods

The experimental design and the selection of sampling points in each study field were based on *an priori* apparent soil electrical conductivity (EC_a) (0-0.70 m depth) map. This was derived from an on-the-go survey conducted in 2014 using a global navigation satellite systems (GNSS) enabled DualEM-1s sensor (Milton, ON, Canada) (Fig. 1). For consistency and to remove variability between the samples due to textural variation and relative EC_a signal response, three sampling points per field were selected under the following criteria;

- The location had an EC_a value of between 8-10 mS m^{-1},
- The location was at least 50 m away from another within field sample site,
- It was not located near the field border (> 20 m from a field boundary), and
- It was not located in an area likely to be disproportionately affected by compaction from either machinery or animal activity.

Across the 12 selected study fields, there were 36 sampling points (2 agricultural systems: 6 fields per system: 3 replicates per study field). At each point, two undisturbed soil cores (0-0.90 m depth) were collected using a hydraulic soil sampler (Atlas Copco Ltd., Hemel Hempstead, Hertfordshire, UK) and a metallic tube (1 m length, 0.03 m inner diameter), totalling 72 sampled cores across the farm. The soil cores were manually cut during sampling into 0-0.15, 0.15-0.30 and 0.30-0.60 m depths resulting in a total of 216 undisturbed soil core sections. In addition, three disturbed samples (0-0.15 m) were also taken using an auger near each of the 36 sample points to provide 108 disturbed soil samples. Soil sampling was conducted in February-March 2017 and the position of each sampled point was geo-referenced with an EGNOS-enabled handheld GPS receiver (Garmin eTrex® 30x).

2.2.1 SQ indicators, soil preparation and analyses

The following seven SQ indicators were analysed: chemical - active acidity (pH), Olsen's phosphorus (P) and ammonium nitrate-extractable potassium (K); physical - aggregate stability (AS) and bulk density (BD); and biological - soil C concentration (C) and microbial biomass carbon (MBC). These SQ indicators were chosen based on productivity and environmental protection management goals and their influence on critical/supporting soil functions and potential threats. The productivity and environmental protection goals are related to the capacity of the system to enhance or maintain the production quantity, quality and stability as well as its efficiency to improve or maintain soil, air and water quality (Andrews *et al.*, 2004).

Each of the 216 fresh undisturbed samples was gently mixed and passed through a 4 mm sieve; large stones were removed and weighed plant remains were discarded. The weight of the sieved, fresh soil was then recorded. A subsample of the sieved soil (5 g) was used for determination of gravimetric water content. BD was calculated using the core method adjusting for the weight and volume of large stones (Blake & Hartge, 1986). Thereafter, the duplicate core samples taken at the same georeferenced location and same depth interval were merged and sieved through a 2 mm sieve. This resulted in 108 merged samples, which were then air-dried before being used for particle-size distribution (PSD), pH, P, K, and C.

PSD was determined by a low angle laser light scattering technique (Laser diffraction) using a Malvern Mastersizer 2000 optical bench with recirculating wet cell enhancement and a Hydro 2000MU sample introduction unit. Soil available P concentration was measured by Olsen's P method (Olsen & Sommers, 1982), soil available K was analysed by extraction with NH_4NO_3 (Anon, 1986) and measurement of K concentrations using a flame photometer, and pH was measured in H_2O (1 : 2.5 soil : solution). Soil C concentration was determined by dry combustion, post-combustion and reduction tube in an Elementary Vario Macro Cube analyser (furnace at 960 °C in pure oxygen).

All 108 disturbed soil samples were used for AS and MBC measurements. First, the three samples from the same location point were combined and sieved through a 4 mm mesh to make a composite sample. MBC was assessed using the D glucose respiration rate derived from the MicroResp™ rapid microtiter plate method (Campbell *et al.*, 2003). MBC was calculated from the biomass respiration measurements following procedures described in West & Sparling (1986). The remaining portion of each sample was air-dried and sieved through a 2 mm sieve above a 1 mm sieve. The aggregates collected on the 1 mm sieve (1-2 mm diameter) were used to determine soil AS using a wet-sieving procedure, which measured the effective resistance of the soil structure against either mechanical or physicochemical collapsing forces (Bourget & Kemp, 1957).

2.3 Statistical analyses

Since the study was carried out on a commercial farm with a stratified selection of the sampling points, spatial autocorrelation and heterogeneity were tested computing the Moran's I index and via a likelihood ratio test (LRT) comparing the null model (an intercept-only model) and the additional, nested model containing a random effect associated with each study field. The latter was confirmed and therefore, linear mixed-effects models (LME) were fitted to each individual SQ indicator (pH, P, K, BD, AS, C, and MBC) to test the effects of agricultural systems (S) (conventional-CONV *vs.* organic-ORG), grazing regime (G) (non-grazed-NG *vs.* grazed-GG) and their interaction (S*G). The model structure used S and G, as fixed effects while the random effect was defined as the study field to account for the heterogeneity of the experimental design. The analyses were conducted separately for each depth interval.

LME models were also used to test the effects of ley time proportion (LTP) (i.e. % years under temporary grass-clover leys in 10 years) on each individual indicator (pH, P, K, BD, AS, C, and MBC). In this case, LTP was used as a continuous variable and as a fixed effect, with study fields as a random effect and analysis being performed separately by depth interval. Although not within the objectives of the study, the same approach was performed to assess potential effects of manure application proportion (MAP) (i.e. % years with manure application in 10 years prior to sampling) on each individual SQ indicator.

For all LME models, assumptions were checked for normality and equal variances by examining the QQ plots of residuals (for both fixed and random effects compartments of the model) and scatterplots of standardised against fitted values. The data were Tukey's Ladder of Powers transformed when visual breakdowns in LME model assumptions were revealed by residual plots. The significance of the fixed effects was determined by comparing models with and without the factor of interest using LRT. When the interaction term in the model was significant, Tukey's HSD post-hoc test was carried out and a significant effect was determined at $p < 0.05$. All statistical analysis was carried out in the R programming language 3.4.3 (R Development Core Team, 2018) using the additional packages, ape (Paradis *et al.*, 2004), nlme (Pinheiro. *et al.*, 2018), plyr (Wickham, 2011), ggplot2 (Wickham, 2009), and multcomp (Hothorn *et al.*, 2008).

3. Results

The data did not show spatial autocorrelation for any of the SQ indicators measured or depth intervals ($p > 0.05$; data not shown), indicating that the sampling strategy based on EC_a analysis (0-0.70 m depth) (Fig. 1) was effective. Agricultural systems (S) (conventional-CONV *vs.* organic-ORG) associated with grazing regimes (G) (non-grazed-NG *vs.* grazed-GG) and LTP (i.e. % years under temporary grass-clover leys in 10 years) affected soil indicator measurements differently at each depth interval (Table 2 and Fig. 2 and 3).

In terms of chemical indicators, pH was not affected by S or G at any soil depth interval ($p>0.05$). For the 0-0.15 m depth, the ORG system showed lower soil P concentration compared to the CONV system (LRT = 10.53; $p = 0.001$, Table 2), while the GG regime significantly increased soil P concentration under both S (LRT= 5.18; $p = 0.02$, Table 2). For the 0.15-0.30 and 0.30-0.60 m depth intervals, there was no significant statistical effect of S or G on P concentration (Table 2, $p>0.05$). In the topsoil (0-0.15 m), S and G interacted, resulting in an increased soil K concentration with the combination of the ORG system and the GG regime (LRT = 4.25; $p = 0.04$, Fig. 2a), while the GG regime had no effect on soil K concentration under the CONV system. Soil K concentration was lower under the GG regimes at 0.15-0.30 m soil depth (LRT = 10.35; $p = 0.001$, Table 2) and was higher in the CONV system at 0.30-0.60 m soil depth (LRT = 5.00; $p = 0.02$, Table 2).

For the physical indicators, an interactive effect between S and G was found for soil BD in the 0-0.15 and 0.30-0.60 m layers. The GG regime under the CONV system decreased BD at 0-0.15m (LRT = 5.66; $p = 0.02$, Fig. 2b), while the GG regime under the ORG system increased BD at 0.30-0.60 m (LRT = 4.04; $p = 0.04$, Fig. 2c) relative to NG. The S and G did not affect AS ($p>0.05$), even though the GG fields showed approximately 10% higher AS on average relative to the NG fields for the 0-0.15 m depth.

For the biological indicators, soil C concentration was higher under the GG regime in the 0-0.15 m depth (LRT = 9.10; p = 0.003, Table 2). There was an interaction between S and G, indicating that the GG regime increased soil C concentration under the CONV system in the 0.15-0.30 m depth interval (LRT = 4.89; p = 0.03, Fig. 2d), but had no effect in the ORG system. The CONV system showed higher soil C concentration in the deeper soil layers (0.30-0.60 m) compared to the ORG system (LRT = 6.48; p = 0.01). The ORG system showed higher soil MBC concentration compared to the CONV system (LRT = 4.23; p = 0.04). The GG regime also significantly increased MBC concentration for the 0-0.15 m depth interval under both S (LRT = 4.19; p = 0.04).

The effects of S (CONV *vs.* ORG), G (NG *vs.* GG) and their interactions (S*G) were also assessed on SQ indicators across the whole soil profile (0-0.60 m) (Table S1). Most of the findings reflected those found for the top 0-0.15 m depth interval, except for the soil K and C concentrations that showed no S or G effects when the whole soil profile was considered. This demonstrates the benefit of individually assessing separate depth intervals as some effects might be masked when soil layers are combined.

Increased LTP did not affect soil pH, P, BD and MBC at any depth interval studied ($p>0.05$, Fig. 3). There was a trend towards increased topsoil K and MBC concentration (0-0.15 m) as LTP increased. An increased LTP significantly increased AS in the 0-0.15 m depth (p = 0.05) and soil C concentration in the 0-0.15 m and 0.15-0.30 m depth (p = 0.002, p = 0.05, respectively). In contrast, as LTP increased, soil K concentration decreased in the 0.15-0.30 m depth (p = 0.007 (Fig. 3). MAP (i.e. % years with manure application in 10 years) did not affect any of the soil indicators measured (pH, P, K, BD, AS, C and MBC) at any of the three depth intervals (0-0.15; 0.15-0.30 and 0.30-0.60 m) assessed (data not shown).

4. Discussion

4.1 Effects of an organic system on SQ indicators

The lower soil available P concentration in the topsoil (0-0.15 m) in the organic system reflected other studies which have reported challenges with maintaining topsoil available P in organic cropping systems (Goulding *et al.*, 2009; Løes & Ebbesvik, 2017; Cooper *et al.*, 2018). Løes & Ebbesvik, (2017) reported that topsoil available P concentration (0-0.20 m) can decrease by half after conversion from a conventional to an organic system. Cooper *et al.* (2018), in a recent survey across Europe, found a declining trend in the soil available P concentrations under organic systems. The decrease in soil available P in organic systems is often associated with an imbalance between the export of P in products and the import of nutrients in livestock feed or approved fertilisers. This imbalance can jeopardise nutient cycling function and reduce the capacity of the organic systems to deliver ecosystem services, such as biomass production in the long-term (Goulding *et al.*, 2009; Cooper *et al.*, 2018). However, it is also possible that the Olsen's P test does not accurately assess the pool of available P in the organically managed soils (Kratz *et al.*, 2016; Cooper *et al.*, 2018). The broad range of elements provided by organic amendments might have caused sorption of P or immobilization in microbial biomass; these forms of P may be slowly available to crops but not reflected in the results of the Olsen's P test (Möller *et al.*, 2018). In addition, the significantly higher MBC in the organic system should reflect a higher level of microbial activity with increased capacity to mobilise nutrients from inaccessible pools including organic P and sorbed P (Maeder *et al.*, 2002).

The absence of a difference between the conventional and organic system in the topsoil (0-0.30 m) K concentration can be explained by the fact that FYM, used as a source of K fertiliser in the organic system, is providing an equivalent supply of K to conventional K fertilisers (Fortune *et al.*, 2006). Nonetheless, differences in soil K concentrations deeper in the soil profile (> 0.30 m) between conventional and organic systems are rarely examined in the literature. Alfaro *et al.* (2006) investigated the effects of N application and drainage of K in grasslands and found higher K leaching as N application was increased. This was attributed to the acidification of the topsoil by synthetic N fertilisers and displacement of cations (including K) on the exchange complex, leading to K leaching down the profile. This could be a mechanism to explain the elevated concentration of K in the conventionally managed subsoils (0.30-0.60 m) and the lower values in the topsoil, relative to the organic. The sustained levels of K in the topsoil in organically managed soils indicate effective nutrient retention, possibly on the cation exchange complex which may be enhanced by the FYM additions.

The higher MBC under the organic system is in agreement with a recent global meta-analysis conducted by Lori *et al.* (2017), who observed a positive effect on soil microbial community abundance and activities when fields are managed organically. The authors pointed out that organic amendments and a more diverse rotation, particularly with the inclusion of legumes, increased the abundance of the microbial community. In this study, conventional and organic inputs and to a certain extent rotation system were alike, but only the organic part of the farm had the inclusion of nitrogen-fixing beans, whereas oilseed rape was only cropped in the conventional system. Although the conventional part of the farm also received organic fertiliser application (FYM), it was used together with mineral fertilisation, which might have affected the efficiency and/or community composition of the microbial biomass (García-Palacios *et al.*, 2018). This theory is also confirmed by the results of Maeder *et al.* (2002), who found enhanced microbial biomass in organically managed soils even when compared to the conventional system that used mineral fertiliser plus FYM.

Previous research has reported that organic systems can also increase topsoil (< 0.20 m depth) C concentrations (Marriott & Wander, 2006; Scialabba & Müller-Lindenlauf, 2010; Gattinger *et al.*, 2012), with very limited studies assessing deeper layers (Blanco-Canqui *et al.*, 2017). In this study, soil C concentrations in the topsoil layers (i.e. 0-0.15 m and 0.15-0.30 m) were not affected while concentrations were lower under the organic system at the 0.30-0.60 m depth interval. Previous research has attributed higher soil C concentrations in organic systems to higher C inputs (through manure, slurry and/or compost application) (Leifeld & Fuhrer, 2010; Gattinger *et al.*, 2012; Kirchmann *et al.*, 2016), but in this study, both conventional and organic systems had regular applications of FYM, as well as ley periods in the rotation, which might have limited differences between the two systems in the topsoil layers. Moreover, it is worth noting that changes in soil C occur slowly (Smith *et al.*, 2020), and therefore the short period since conversion to the organic system (~ 15 years) may have not allowed for detectable changes.

The significantly higher soil C concentration at 0.30-0.60 m depth under the conventional system contradicted previous work. Blanco-Canqui *et al.* (2017), in a long-term experiment (+20 years), did not find significant differences in soil C concentrations between a conventional and an organic system below 0.15 m depth, but they highlighted that in the organic system there was a trend towards higher soil C concentrations with the implementation of a more diversified rotation treatment and deep-rooting crops. However, studies comparing soil properties in deeper soil profiles between organic and non-organic systems are limited. In this study, the typically large aboveground biomass in the conventional system should equate to a larger belowground biomass (Bilsborrow *et al.*, 2013). This could have resulted in a larger, deeper rooting system under the conventionally managed soils that enhanced soil C concentrations in the deeper (0.30-0.60 m) layer. This finding has implications for the climate regulation function of soils. While organic systems are commonly reported to have less of an impact on climate due to lower emissions from fertiliser manufacture (Smith *et al.*, 2019), increasing C concentrations in deeper soil layers could result in increased C sequestration at depth, which may partially offset GHG emissions from conventional systems (Tautges *et al.*, 2019).

Organic systems have been reported to trigger beneficial feedback loops between plants and microbial biomass that ultimately stimulates the plant to promote its own microbial population to increase nutrient availability and utilisation from organic material (Hamilton & Frank, 2001; Stockdale *et al.*, 2006). This is facilitated by microbial exudates, which would also bring further long-term benefits to soil aggregation and to soil C quantity and stability (Tisdall & Oades, 1982; Loaiza Puerta *et al.*, 2018). In this regard, it was expected that soil physical properties (i.e. BD and AS) would be enhanced in organic systems. Where soil type is the same, differences in physical properties such as BD and AS are largely driven by soil C contents. In this study, since soil type and soil C contents were similar for both systems, it is not surprising that AS and BD were also not significantly different when comparing the two systems. This suggests that the soil functions linked to soil structure, including regulation of the water cycle and provision of physically stable aggregates, do not differ between conventional and organic systems.

Overall, the potentially higher organic and microbial forms of P, similar topsoil (0-0.30 m) K, BD, AS and C concentration and the higher MBC under the organic system indicate that agricultural systems receiving only organic amendments and including nitrogen-fixing plants in the rotation can generate analogous SQ with fewer external inputs than conventional systems.

4.2 Effects of the grazing regime and its interaction with agricultural systems on SQ indicators

The higher topsoil (0-0.15 m) available P, C and MBC under grazed regimes (compared to non-grazed) were likely to be associated with the higher nutrient returns and enhanced nutrient cycling provided by animals, ley periods and residues left in the soil.

Topsoil (0-0.15 m) available P was 40% and 240% higher under conventional and organic grazed regimes respectively, when compared with non-grazed counterparts (Table 2). According to Nash *et al.* (2014), up to 85% of the P applied and taken up by plants is returned to the soil via animal dung in a grazed system. Since animals in a grazed regime act as a nutrient cycling agent (Carvalho *et al.*, 2010), it is likely that they modify both the biochemical form of the nutrients and their spatial distribution, and consequently influence local availability in the soil solution. Moreover, grazing can change plant population dynamics and species diversity, resulting in a different plant ecology system compared to a non-grazed regime (Assmann *et al.*, 2017). This increased soil P availability effect can be found even under light grazing intensities (Assmann *et al.*, 2017) and has been observed across varying mixed (crop-livestock) production systems in Europe (Cooper *et al.*, 2018). However, studies directly comparing conventional and organic mixed farming systems in association with non-grazed and grazed regimes, as compared in this study, are rare (Jackson *et al.*, 2019). This finding on soil available P merits particular attention for future discussions on sustainable agriculture strategies as mineral P (as rock phosphate) is a finite resource. Increased available P under organic grazed regimes suggests that grazing residues (urine and dung) and organic amendments are complementary strategies (Assmann *et al.*, 2017) which may be beneficial for cropping systems at a lower level of P supply.

The grazed regime also increased topsoil (0-0.15 m) C concentration and MBC under both agricultural systems (Table 2). Previous studies have also found that implementing grazing can increase topsoil C concentration (Abdalla *et al.*, 2018), indicating that the soil C gains may be limited to the surface layers where the root systems dominate (Medina-Roldán *et al.*, 2008; Chen *et al.*, 2015). Increased MBC in grazed fields might be related to interlinked mechanisms regarding the effects of grazing on the microbial community, including changes in biomass production and resource allocation, resource inputs to the decomposers and the plant community itself (Bardgett & Wardle, 2003). Together, these suggest that grazing could be driving soil C accumulation and MBC in the top 0-0.15 m depth due to greater deposition of easily available C inputs and nutrients, which indirectly stimulates below-ground biomass (e.g. root growth), followed by greater root turnover and exudations (McSherry & Ritchie, 2013; Chen *et al.*, 2015).

Grazing intensity may influence soil C concentration and MBC positively or negatively by changing individual plant species and plant cover as well as processes that fix C during photosynthesis as a function of microclimate (McSherry & Ritchie, 2013; Abdalla *et al.*, 2018).

Since in our study grazing intensity was relatively low and climate parameters were similar for all study fields, the residue amount left in the soil by animals and root growth are likely to be the primary causes of the higher C concentration and MBC in the grazed regimes. We hypothesise that animal trampling may have incorporated part of the residues deposited on the soil surface into the topsoil, whilst also stimulating greater root growth and turnover. These mechanisms could be especially important for the 0.15-0.30 m depth in the conventional system, which showed the lowest soil C concentration in non-grazed fields but a significant increase in grazed regimes (Table 2 and Fig. 2). Lower soil C concentration in conventional non-grazed study fields may also be related to the use of more mineral N fertiliser and an increase in residue decomposability (García-Palacios *et al.*, 2018). While grazed regimes have increased topsoil (0-0.15 m) C concentration and MBC, grazing ruminants on leys results in GHG emissions and reduces land available for cereal crop production. This illustrates the complexity of decision making about land management practices once the multiple ecosystem services provided by agricultural landscapes are considered. Further research is required to assess the trade-offs between the C sequestration benefits of grazed leys and the wider impacts on the food system.

The grazed regime also interacted with agricultural system enhancing topsoil (0-0.15 m) K concentration under the organic system (Table 2 and Fig. 2). Grazed organic systems experience high degree of recycling of K through the return of dung, especially urine, since only a small portion of K is retained in animal products (e.g. milk and meat) (Haynes & Williams, 1993; Assmann *et al.*, 2017). This cycling of K, in combination with higher rates of FYM inputs on organic fields (averages of 100 and 166 kg K ha^{-1} yr^{-1}, for the conventional and organic system in the last 10 years, respectively) could result in high levels of available K in grazed organic fields.

In contrast, the non-grazed regime showed nearly twice as much available K in the 0.15-0.30 m compared to the grazed fields regardless of the agricultural system. This corresponds to results from a review conducted in Brazil by de Faccio Carvalho *et al.* (2010) who found that non-grazed fields have higher K concentrations in the soil profile, in particular from 0.10 to 0.30 m soil depth. The main hypothesis for the higher K concentration in the non-grazed field at depth is that grazed fields possess a denser root system in the topsoil that mines subsurface K reserves (0.15-0.30 m) and recycles and deposits this K onto the soil surface (0-0.15 m). However, more research on the morphology of ley root systems under non-grazed and grazed regime is required to further elucidate these mechanisms.

Changes in root growth quantity and dynamics might also explain the interactive effect found in soil BD. The decrease in topsoil (0-0.15 m) BD in conventional grazed fields, compared to conventional non-grazed fields, may be linked to the stimulation of root growth resulting in an increase in the root exudation and microbial activities (confirmed by our MBC results and also by Hamilton & Frank, 2001). In organic systems, the higher nutrient availability in the surface layers under grazed fields (Table 2) may have discouraged the need for root development into the deeper soil layers, resulting in a higher BD for 0.30-0.60 m depth. A potential stimulation of surface below-ground biomass production by grazing is an important feature as it can amplify the formation of soil aggregates and reduce soil compaction (Dominy & Haynes, 2002). Although not significant ($p = 0.09$, Table 2), soil aggregate stability was 10% higher in the topsoil of grazed fields compared to non-grazed fields and appeared to be linked to the length of time that a field was in the ley phase (see section 4.3). This indicates that important soil functions, including mitigation of GHG emissions (Ball, 2013), resistance to soil erosion (Barthès & Roose, 2002), and improved water infiltration and retention, may all be enhanced by grazed ley periods. Our results, therefore, indicate an enhanced SQ from mixed farming systems that could have potential policy implications for the design of multifunctional landscapes.

4.3 Effects of ley time proportion (LTP) on SQ indicators

Increasing LTP in the crop rotation increased AS (0-0.15 m) and C concentration (0-0.15 and 0.15-0.30 m) under both agricultural systems, while it decreased K concentration in the 0.15-0.30 m depth (Fig. 3). The decreased soil K concentration at this intermediate-depth interval with increased LTP, supports the notion that a more extensive root system might be mining K from the 0.15-0.30 m depth and depositing it onto the soil surface (0-0.15 m); the trend (non-significant) towards increased topsoil K (0-0.15 m) as LTP increased further supports this hypothesis. The development of a dense root system may also lead to improved soil aggregate stability (i.e. soil structure), and favour the protection and stabilisation of SOM as well as associated nutrients (Six *et al.*, 2002). This is supported by the observed increased AS (0-0.15 m) and soil C concentration (0-0.15 and 0.15-0.30 m) with increased LTP.

The results of this study agree with findings from other studies assessing the effects of LTP on soil structure and soil C concentration (Jarvis *et al.*, 2017; Loaiza Puerta *et al.*, 2018; Crème *et al.*, 2018). Jarvis *et al.* (2017) compared varying proportions of ley (1, 2, 3 or 5 years) in a long-term field trial (60 years) and found that higher proportions of ley time in a rotation improved both topsoil structure and C concentration. Similarly, Loaiza Puerta *et al.* (2018)

reported improved soil aggregate stability and soil C concentration after two years following four years of arable cropping. Crème *et al.* (2018) assessed the legacy effect of 3 and 6 years of grassland ley periods after 3 years arable cropping and found that even under short periods (i.e. 3 years) the soil C concentration increased with the implementation of ley periods compared to continuous arable production.

Most previous studies have indicated higher soil aggregate stability and C concentration in a ley-arable rotation compared to continuous arable in the topsoil layers (max. 0.20 m soil depth). This study supports these findings, but also reported increased soil C concentration for intermediate soil layers (i.e. 0.15-0.30 m), which is a significant outcome. In one of the few studies assessing the effects of ley-arable rotations on soil C below 0.20 m, Blanco-Canqui *et al.* (2017) found no significant effect below 0.15 m soil depth. The authors considered two-year ley periods in a four-year crop rotation, concluding that the time under ley (i.e. two years) was insufficient to develop an extensive and deep root system to build soil C concentration in the subsoil. Our results suggest that grass-clover ley for approximately 30-40% of the crop rotation (i.e. 3-4 years in a 10-year period) may be required to increase C concentration at 0.15-0.30 m depth. This is particularly relevant for future policies relating to climate change mitigation since building soil C in deeper layers can result in slower rates of decomposition and improve C protection and sequestration in the soil (Lorenz & Lal, 2005). Increasing LTP has increased AS (0-0.15 m) and C concentration (0-0.15 and 0.15-0.30 m) and its wide adoption to improve SQ could result in a return to mixed farming systems and less specialisation of crop or livestock farms. This could have GHG implications if total ruminant numbers increased, something that would need investigation using a life-cycle assessment approach to point out the real benefits and/or drawbacks of different scenario.

5. Conclusions

This research was performed in commercial mixed farm in northern England to investigate the impacts of organic and non-organic (conventional) agricultural systems on soil quality (SQ) indicators in both the topsoil and subsoil. More specifically, it investigated how changes from a conventional to an organic system and the presence (or absence) of grazing regimes (non-grazed *vs.* grazed) and pasture leys in rotation, and their interactions, influenced chemical, physical, and biological soil quality indicators. For the topsoil, the findings reflected existing knowledge on the advantages of organic *vs.* conventional systems on SQ indicators. When grazing was included, both agricultural systems benefited from a greatly enhanced SQ, in particular the grazed conventional system. The grazed organic system had a much smaller benefit compared to the non-grazed organic system. The length of pasture leys in the rotation was positively related to SQ regardless of the type of agricultural system, and a grass-clover ley period length equivalent to 30-40% of the full crop rotation is needed to increase AS and soil C concentration in a linear fashion. Subsoil conditions (below 0.30 m) showed a different pattern for SQ to the topsoil. Bulk density and soil C accumulation were favoured under the conventional system, which is hypothesised to be due to a larger and deeper rooting system. Studies into subsoil SQ indicators are less common and the results here show that the agricultural system effects are probably more complex than in the topsoil. However, including grazing and pasture leys in management systems has positive benefits throughout the profile on SQ indicators regardless of whether the system is conventionally or organically managed. Ultimately, reviving mixed farming systems may be a key factor for delivering multi-functional agroecosystems that maintain SQ and optimise ecosystem services including nutrient recycling/release and utilisation. This still needs more research, particularly in furthering knowledge of how subsoil SQ indicators respond to management and also on economic considerations of any proposed changes in management.

Acknowledgements

This work was supported by the Faculty of Science, Agriculture & Engineering, Newcastle University [SAgE Scholarship]. The authors also thank the MSc graduates Pengliang Shang, Ayobami Oladipo and Sarah Wyld for their contributions with some soil analyses and Gavin Hall and Rachel Chapman for their assistance in the soil sampling campaign. The authors of this study declare no conflict of interest. The data that support the findings of this study are available in both in the main body of the paper and in the supplementary material of this article. Raw data can also be provided on request from the corresponding author. The authors also thanked the two anonymous reviewers for their valuable criticisms and comments, which led to substantial improvements to the manuscript.

References

Abdalla, M., Hastings, A., Chadwick, D.R., Jones, D.L., Evans, C.D., Jones, M.B., Rees, R.M. & Smith, P. 2018. Critical review of the impacts of grazing intensity on soil organic carbon storage and other soil quality indicators in extensively managed grasslands. *Agriculture, Ecosystems & Environment*, **253**, 62–81.

Alfaro, M.A., Jarvis, S.C. & Gregory, P.J. 2006. Potassium budgets in grassland systems as affected by nitrogen and drainage. *Soil Use and Management*, **19**, 89–95.

Amundson, R., Berhe, A.A., Hopmans, J.W., Olson, C., Sztein, A.E. & Sparks, D.L. 2015. Soil science. Soil and human security in the 21st century. *Science*, **348**, 1261071.

Andrews, S.S., karlen, D.L. & Cambardella, C.A. 2004. The Soil Management Assessment Framework: A quantitative soil quality evvaluation method. *Soil Science Society of America Journal*, **68**, 1945–1962.

Anon. 1986. The analysis of agricultural materials. Ministry of Agriculture Fisheries and Food Reference Book 427, HMSO London.

Assmann, J.M., Martins, A.P., Anghinoni, I., de Oliveira Denardin, L.G., de Holanda Nichel, G., de Andrade Costa, S.E.V.G., Pereira e Silva, R.A., Balerini, F., de Faccio Carvalho, P.C. & Franzluebbers, A.J. 2017. Phosphorus and potassium cycling in a long-term no-till integrated soybean-beef cattle production system under different grazing intensities insubtropics. *Nutrient Cycling in Agroecosystems*, **108**, 21–33.

Ball, B.C. 2013. Soil structure and greenhouse gas emissions: A synthesis of 20 years of experimentation. *European Journal of Soil Science*, **64**, 357–373.

Bardgett, R.D. & Wardle, D.A. 2003. Herbivore-Mediated Linkages between Aboveground and Belowground Communities. *Ecology*, **84**, 2258–2268.

Barthès, B. & Roose, E. 2002. Aggregate stability as an indicator of soil susceptibility to runoff and erosion; validation at several levels. *Catena*, **47**, 133–149.

Bilsborrow, P., Cooper, J., Tétard-Jones, C., Średnicka-Tober, D., Barański, M., Eyre, M., Schmidt, C., Shotton, P., Volakakis, N., Cakmak, I., Ozturk, L., Leifert, C. & Wilcockson, S. 2013. The effect of organic and conventional management on the yield and quality of wheat grown in a long-term field trial. *European Journal of Agronomy*, **51**, 71–80.

Blake, G.H. & Hartge, K.H. 1986. *"Bulk density", in Methods of soil analysis.* (A Klute, Ed.). 2nd ed. The American Society of Agronomy.

Blanco-Canqui, H., Francis, C.A. & Galusha, T.D. 2017. Does organic farming accumulate carbon in deeper soil profiles in the long term? *Geoderma*, **288**, 213–221.

Bourget, S.J. & Kemp, J.G. 1957. Wet sieving apparatus for stability analysis of soil aggregates. *Canadian Journal of Soil Science*, **37**, 60–61.

Bünemann, E.K., Bongiorno, G., Bai, Z., Creamer, R.E., De Deyn, G., de Goede, R., Fleskens, L., Geissen, V., Kuyper, T.W., Mäder, P., Pulleman, M., Sukkel, W., van Groenigen, J.W. & Brussaard, L. 2018. Soil quality – A critical review. *Soil Biology and Biochemistry*, **120**, 105–125.

Campbell, C.D., Chapman, S.J., Cameron, C.M., Davidson, M.S. & Potts, J.M. 2003. A Rapid Microtiter Plate Method To Measure Carbon Dioxide Evolved from Carbon Substrate Amendments so as To Determine the Physiological Profiles of Soil Microbial Communities by Using Whole Soil. *Applied and Environmental Microbiology*, **69**, 3593–3599.

Carvalho, J.L.N., Raucci, G.S., Cerri, C.E.P., Bernoux, M., Feigl, B.J., Wruck, F.J. & Cerri, C.C. 2010. Impact of pasture, agriculture and crop-livestock systems on soil C stocks in Brazil. *Soil and Tillage Research*, **110**, 175–186.

Chen, W., Huang, D., Liu, N., Zhang, Y., Badgery, W.B., Wang, X. & Shen, Y. 2015. Improved grazing management may increase soil carbon sequestration in temperate steppe. *Scientific Reports*, **5**, 10892.

Connor, D.J. 2008. Organic agriculture cannot feed the world. *Field Crops Research*, **106**, 187–190.

Cooper, J., Reed, E.Y., Hörtenhuber, S., Lindenthal, T., Løes, A.-K., Mäder, P., Magid, J., Oberson, A., Kolbe, H. & Möller, K. 2018. Phosphorus availability on many organically managed farms in Europe. *Nutrient Cycling in Agroecosystems*, **110**, 227–239.

Cranfield University. 2020. The Soils Guide. *Cranfield University, UK.*, (At: www.landis.org.uk. Accessed: 5/4/2020).

Crème, A., Rumpel, C., Le Roux, X., Romian, A., Lan, T. & Chabbi, A. 2018. Ley grassland under temperate climate had a legacy effect on soil organic matter quantity, biogeochemical signature and microbial activities. *Soil Biology and Biochemistry*, **122**, 203–210.

Dominy, C.S. & Haynes, R.J. 2002. Influence of agricultural land management on organic matter content, microbial activity and aggregate stability in the profiles of two Oxisols. *Biology and Fertility of Soils*, **36**, 298–305.

Doran, J.W. 2002. Soil health and global sustainability: Translating science into practice. In: *Agriculture, Ecosystems and Environment*, pp. 119–127.

de Faccio Carvalho, P.C., Anghinoni, I., de Moraes, A., de Souza, E.D., Sulc, R.M., Lang, C.R., Flores, J.P.C., Terra Lopes, M.L., da Silva, J.L.S., Conte, O., de Lima Wesp, C., Levien, R., Fontaneli, R.S. & Bayer, C. 2010. Managing grazing animals to achieve nutrient cycling and soil improvement in no-till integrated systems. *Nutrient Cycling in Agroecosystems*, **88**, 259–273.

Farewell, T.S., Truckell, I.G., Keay, C.A. & Hallett, S.H. 2011. *The derivation and application of Soilscapes: soil and environmental datasets from the National Soil Resources Institute, Cranfield University*.

Fortune, S., Robinson, J.S., Watson, C.A., Philipps, L., Conway, J.S. & Stockdale, E.A. 2006. Response of organically managed grassland to available phosphorus and potassium in the soil and supplementary fertilization: field trials using grass–clover leys cut for silage. *Soil Use and*

Management, **21**, 370–376.

García-Palacios, P., Gattinger, A., Bracht-Jørgensen, H., Brussaard, L., Carvalho, F., Castro, H., Clément, J.-C., De Deyn, G., D'Hertefeldt, T., Foulquier, A., Hedlund, K., Lavorel, S., Legay, N., Lori, M., Mäder, P., Martínez-García, L.B., Martins da Silva, P., Muller, A., Nascimento, E., Reis, F., Symanczik, S., Paulo Sousa, J. & Milla, R. 2018. Crop traits drive soil carbon sequestration under organic farming. *Journal of Applied Ecology*, **55**, 2496–2505.

Gattinger, A., Muller, A., Haeni, M., Skinner, C., Fliessbach, A., Buchmann, N., Mäder, P., Stolze, M., Smith, P., Scialabba, N.E.-H. & Niggli, U. 2012. Enhanced top soil carbon stocks under organic farming. *Proceedings of the National Academy of Sciences*, **109**, 18226–18231.

Godfray, H.C., Beddington, J.R., Crute, I.R., Haddad, L., Lawrence, D., Muir, J.F., Pretty, J., Robinson, S., Thomas, S.M. & Toulmin, C. 2010. Food security: the challenge of feeding 9 billion people. *Science*, **327**, 812–818.

Gomiero, T., Pimentel, D. & Paoletti, M.G. 2011. Environmental Impact of Different Agricultural Management Practices: Conventional vs. Organic Agriculture. *Critical Reviews in Plant Sciences*, **30**, 95–124.

Goulding, K., Stockdale, E. & Watson, C. 2009. Plant Nutrients in Organic Farming. In: *Organic Crop Production – Ambitions and Limitations*, pp. 73–88. Springer Netherlands, Dordrecht.

Hamilton, E.W. & Frank, D.A. 2001. Can plants stimulate soil microbes and their own nutrient supply? *Ecology*, **82**, 2397–2402.

Haynes, R.J. & Williams, P.H. 1993. Nutrient Cycling and Soil Fertility in the Grazed Pasture Ecosystem. *Advances in Agronomy*, **49**, 119–199.

Hothorn, T., Bretz, F. & Westfall, P. 2008. Simultaneous Inference in General Parametric Models. **50**, 346–363.

IFOAM. 2012. *The IFOAM Norms for Organic Production and Processing* (B (International Federation of Organic Agriculture Movements (IFOAM) and 2012) Germany, Eds.).

Jackson, R.D., Isidore, B. & Cates, R.L. 2019. Are plant-soil dynamics different in pastures under organic management? A review. *Agriculture, Ecosystems and Environment*, **279**, 53–57.

Jarvis, N., Forkman, J., Koestel, J., Kätterer, T., Larsbo, M. & Taylor, A. 2017. Long-term effects of grass-clover leys on the structure of a silt loam soil in a cold climate. *Agriculture, Ecosystems & Environment*, **247**, 319–328.

Johnston, A.E., Poulton, P.R., Coleman, K., Macdonald, A.J. & White, R.P. 2017. Changes in soil organic matter over 70 years in continuous arable and ley-arable rotations on a sandy loam soil in England. *European Journal of Soil Science*, **68**, 305–316.

Karlen, D.L., Andrews, S.S., Wienhold, B.J. & Zobeck, T.M. 2008. Soil Quality Assessment: Past, Present and Future . *Journal of Integrative Biosciences* , **6**, 3–14.

Karlen, D.L., Mausbach, J.W., Doran, R.G., Cline, R.G., Harris, R.F. & Schuman, G.E. 1997. Soil Quality: A Concept, Definition, and Framework for Evaluation. *Soil Science Society of America Journal*, **61**, 4–10.

Kirchmann, H., Kätterer, T., Bergström, L., Börjesson, G. & Bolinder, M.A. 2016. Flaws and criteria for design and evaluation of comparative organic and conventional cropping systems. *Field Crops Research*, **186**, 99–106.

Kratz, S., Schick, J. & Øgaard, A.F. 2016. P Solubility of Inorganic and Organic P Sources. In: *Phosphorus in Agriculture: 100 % Zero* (eds. Schnug, E. & De Kok, L.J.), pp. 127–154. Springer Netherlands, Dordrecht.

Lal, R. 2004. Soil carbon sequestration impacts on global climate change and food security. *Science*, **304**, 1623–1627.

Lal, R. 2007. Carbon Management in Agricultural Soils. *Mitigation and Adaptation Strategies for Global Change*, **12**, 303–322.

Leifeld, J. & Fuhrer, J. 2010. Organic farming and soil carbon sequestration: what do we really know about the benefits? *Ambio*, **39**, 585–99.

Loaiza Puerta, V., Pujol Pereira, E.I., Wittwer, R., van der Heijden, M. & Six, J. 2018. Improvement of soil structure through organic crop management, conservation tillage and grass-clover ley. *Soil and Tillage Research*, **180**, 1–9.

Løes, A.-K. & Ebbesvik, M. 2017. Phosphorus deficits by long-term organic dairy farming? In: *Innovative research for Organic Agriculutre 3.0. Proceeedings of the Scientific Track, Organic World Congress, ISOFAR, TIPI and NCOF*, pp. 531–534. India.

Lorenz, K. & Lal, R.B.T.-A. in A. 2005. The Depth Distribution of Soil Organic Carbon in Relation to Land Use and Management and the Potential of Carbon Sequestration in Subsoil Horizons. pp. 35–66. Academic Press.

Lori, M., Symnaczik, S., Mäder, P., De Deyn, G. & Gattinger, A. 2017. Organic farming enhances soil microbial abundance and activity—A meta-analysis and meta-regression. *PLOS ONE*, **12**, e0180442.

Maeder, P., Fliessbach, A., Dubois, D., Gunst, L., Fried, P. & Niggli, U. 2002. Soil Fertility and Biodiversity in Organic Farming. *Science*, **296**, 1694–1697.

Marriott, E.E. & Wander, M.M. 2006. Total and labile soil organic matter in organic and conventional farming systems. *Soil Science Society of America Journal*, **70**, 950–959.

McSherry, M.E. & Ritchie, M.E. 2013. Effects of grazing on grassland soil carbon: a global review. *Global Change Biology*, **19**, 1347–1357.

Medina-Roldán, E., Arredondo, J.T., Huber-Sannwald, E., Chapa-Vargas, L. & Olalde-Portugal, V. 2008. Grazing effects on fungal root symbionts and carbon and nitrogen storage in a shortgrass steppe in Central Mexico. *Journal of Arid Environments*, **72**, 546–556.

Meier, M.S., Stoessel, F., Jungbluth, N., Juraske, R., Schader, C. & Stolze, M. 2015. Environmental impacts of organic and conventional agricultural products – Are the differences captured by life cycle assessment? *Journal of Environmental Management*, **149**, 193–208.

Möller, K., Oberson, A., Bünemann, E.K., Cooper, J., Friedel, J.K., Glæsner, N., Hörtenhuber, S., Løes, A.K., Mäder, P., Meyer, G., Müller, T., Symanczik, S., Weissengruber, L., Wollmann, I. & Magid, J. 2018. Improved Phosphorus Recycling in Organic Farming: Navigating Between Constraints. In: *Advances in Agronomy*, pp. 159–237.

Mondelaers, K., Aertsens, J. & Van Huylenbroeck, G. 2009. A meta-analysis of the differences in environmental impacts between organic and conventional farming. *British Food Journal*, **111**, 1098–1119.

Nash, D.M., Haygarth, P.M., Turner, B.L., Condron, L.M., McDowell, R.W., Richardson, A.E., Watkins, M. & Heaven, M.W. 2014. Using organic phosphorus to sustain pasture productivity: A perspective. *Geoderma*, **221–222**, 11–19.

Olsen, S.R. & Sommers, L.E. 1982. *Phosphorus. In: Page AL, Miller RH, Keeney DR (eds) Methods of soil analysis part 2. American Society of Agronomy, Madison, pp 403–430*.

Paradis, E., Claude, J. & Strimmer, K. 2004. APE: analyses of phylogenetics and evolution in R language. Bioinformatics 20: 289-290. R package version 5.0.

Paustian, K., Collins, H.P. & Paul, E.A. 1997. Management controls on soil carbon. In: *Soil organic matter in temperate Agroecosystems - Long-term experiments in North America* (eds. Paul, E.A., Paustian, K., Elliott, E.T., Cole, C. V, Paustian, K., Elliott, E.T. & Cole, C. V), pp. 15–49.

Pickett, J.A. 2013. Food security: intensification of agriculture is essential, for which current tools must be defended and new sustainable technologies invented. *Food and Energy Security*, **2**, 167–173.

Pinheiro., J., Bates., D., DebRoy., S., Sarkar., D. & Team, R.C. 2018. nlme: Linear and Nonlinear Mixed Effects Models. R package version 3.1-131.1. (At: https://cran.r-project.org/package=nlme.).

R Development Core Team. 2018. R: A language and environment to statistical computing. (At: http://www.r-project.org.).

Reay, D.S., Davidson, E.A., Smith, K.A., Smith, P., Melillo, J.M., Dentener, F. & Crutzen, P.J. 2012. Global agriculture and nitrous oxide emissions. *Nature Climate Change*, **2**, 410.

Red Tractor Assurance. 2015. Red Tractor Assurance Standards. *www.Redtractor.Org.Uk*.

Reganold, J.P. & Wachter, J.M. 2016. Organic agriculture in the twenty-first century. **2**, 15221.

Scialabba, N.E.H. & Müller-Lindenlauf, M. 2010. Organic agriculture and climate change. *Renewable Agriculture and Food Systems*, **25**, 158–169.

Seufert, V. & Ramankutty, N. 2017. Many shades of gray—The context-dependent performance of organic agriculture. *Science Advances*, **3**.

Six, J., Feller, C., Denef, K., Ogle, S.M., de Moraes, J.C. & Albrecht, A. 2002. Soil organic matter, biota and aggregation in temperate and tropical soils - Effects of no-tillage. *Agronomie*, **22**, 755–775.

Smith, L.G., Kirk, G.J.D., Jones, P.J. & Williams, A.G. 2019. The greenhouse gas impacts of converting food production in England and Wales to organic methods. *Nature Communications*.

Smith, P., Soussana, J.F., Angers, D., Schipper, L., Chenu, C., Rasse, D.P., Batjes, N.H., van Egmond, F., McNeill, S., Kuhnert, M., Arias-Navarro, C., Olesen, J.E., Chirinda, N., Fornara, D., Wollenberg, E., Álvaro-Fuentes, J., Sanz-Cobena, A. & Klumpp, K. 2020. How to measure, report and verify soil carbon change to realize the potential of soil carbon sequestration for atmospheric greenhouse gas removal. *Global Change Biology*.

Soil Association. 2019. *Soil Association Standards Farming and Growing*. (At: https://www.soilassociation.org/media/15931/farming-and-growing-standards.pdf.).

Stavi, I. & Lal, R. 2012. Agriculture and greenhouse gases, a common tragedy. A review. *Agronomy for Sustainable Development*, **33**, 275–289.

Stockdale, E.A., Shepherd, M.A., Fortune, S. & Cuttle, S.P. 2006. Soil fertility in organic farming systems - fundamentally different? *Soil Use and Management*, **18**, 301–308.

Tautges, N.E., Chiartas, J.L., Gaudin, A.C.M., O'Geen, A.T., Herrera, I. & Scow, K.M. 2019. Deep soil inventories reveal that impacts of cover crops and compost on soil carbon sequestration differ in surface and subsurface soils. *Global Change Biology*.

Tisdall, J.M. & Oades, J.M. 1982. Organic matter and water-stable aggregates in soils. *Journal of Soil Science*, **33**, 141–163.

Tsiafouli, M.A., Thébault, E., Sgardelis, S.P., de Ruiter, P.C., van der Putten, W.H., Birkhofer, K., Hemerik, L., de Vries, F.T., Bardgett, R.D., Brady, M.V., Bjornlund, L., Jørgensen, H.B., Christensen, S., Hertefeldt, T.D., Hotes, S., Gera Hol, W.H., Frouz, J., Liiri, M., Mortimer, S.R., Setälä, H., Tzanopoulos, J., Uteseny, K., Pižl, V., Stary, J., Wolters, V. & Hedlund, K. 2015. Intensive agriculture reduces soil biodiversity across Europe. *Global Change Biology*, **21**, 973–985.

Tuomisto, H.L., Hodge, I.D., Riordan, P. & Macdonald, D.W. 2012. Does organic farming reduce environmental impacts?--a meta-analysis of European research. *J Environ Manage*, **112**, 309–320.

West, A.W. & Sparling, G.P. 1986. Modifications to the substrate-induced respiration method to permit measurement of microbial biomass in soils of differing water contents. *Journal of Microbiological Methods*, **5**, 177–189.

Wickham, H. 2009. ggplot2: Elegant Graphics for Data Analysis. *Springer-Verlag New York. R package version 2.2.1*.

Wickham, H. 2011. The Split-Apply-Combine Strategy for Data Analysis. *Journal of Statistical Software, 40(1), 1-29.*, (At: https://www.jstatsoft.org/v040/i01.).

WRB. 2015. *World Reference Base for Soil Resources 2014, update 2015 International soil classification system for naming soils and creating legends for soil maps. World Soil Resources Reports No. 106. FAO, Rome.*

Table 1. Details of management practices on the 12 study fields at Nafferton Farm over 10 years (2008-2017) indicating agricultural system (conventional and organic), grazing regime (non-grazed and grazed), ley time proportion (LTP) (% years under ley prior sampling) and manure application proportions (MAP) (% years with manure applied prior sampling) in the last 10 years, and further details including main crops grown, fertilisation and tillage occurrence that accounted for any activity that turned the soil over for at least 0.15 m soil depth.

ly field n° in the map	Agricultural system	Grazing regime	LTP %	MAP %	Further details
1	conventional	Non-grazed	0	10	Continuous arable rotation of wheat, barley and oilseed rape crops for the last ten years, eight tillage occurrences. Annual fertilisation (mineral and organic forms) of roughly 89, 78 and 156 kg ha^{-1} yr^{-1} for N, P and K, respectively.
2	conventional	Non-grazed	10	10	Previously cultivated with ley-arable rotation but became a continuous arable rotation of wheat, barley and oilseed rape crops in which the field is for the last nine years, five tillage occurrences. Annual fertilisation (mineral and organic forms) of roughly 69, 56 and 111 kg ha^{-1} yr^{-1} for N, P and K, respectively.
3	conventional	Grazed	70	60	Ley-arable rotation of wheat, barley, three tillage occurrences, and ley in which the field is for the last seven years. Annual fertilisation (mineral and organic forms) of roughly 148, 46 and 93 kg ha^{-1} yr^{-1} for N, P and K, respectively.
4	conventional	Grazed	50	40	Ley-arable rotation of wheat, barley in which the field is for the last four years, four tillage occurrences. Before that, ley was used for five years in a row with one previous year under barley. Annual fertilisation (mineral and organic forms) of roughly 89, 31 and 43 kg ha^{-1} yr^{-1} for N, P and K, respectively.
5	conventional	Grazed	100	50	Ley-arable rotation field but under ley for the last ten years, no tillage occurrence. Annual fertilisation (mineral and organic forms) of roughly 130, 28 and 57 kg ha^{-1} yr^{-1} for N, P and K, respectively.
6	conventional	Grazed	60	40	Ley-arable rotation of wheat, barley, three tillage occurrences, and ley in which the field is for the last four years. Before the ley, the field had three years under arable rotation with the previous three years under ley. Annual fertilisation (mineral and organic forms) of roughly 190, 79 and 140 kg ha^{-1} yr^{-1} for N, P and K, respectively.
7	organic	Grazed	80	60	Ley-arable rotation of wheat, barley, two tillage occurrences, and ley in which the field is for the last seven years. Before the ley, the field had two years under arable rotation and one previous year under ley. Annual fertilisation (only organic forms) of roughly 48, 52 and 141 kg ha^{-1} yr^{-1} for N, P and K, respectively.
8	organic	Grazed	60	70	Ley-arable rotation of wheat, barley, beans, four tillage occurrences, and ley in which the field is for the last four years. Before the ley, the field had three years under arable rotation with the previous two years under ley and one year under beans. Annual fertilisation (only organic forms) of roughly 59, 61 and 150 kg ha^{-1} yr^{-1} for N, P and K, respectively.
9	organic	Grazed	60	20	Ley-arable rotation of barley, beans, potatoes, three tillage occurrences, and ley, which occurred in an interval of every two years of arable crop. Currently, the field is under ley for the last three years. Annual fertilisation (only organic forms) of roughly 59, 65 and 170 kg ha^{-1} yr^{-1} for N, P and K, respectively.

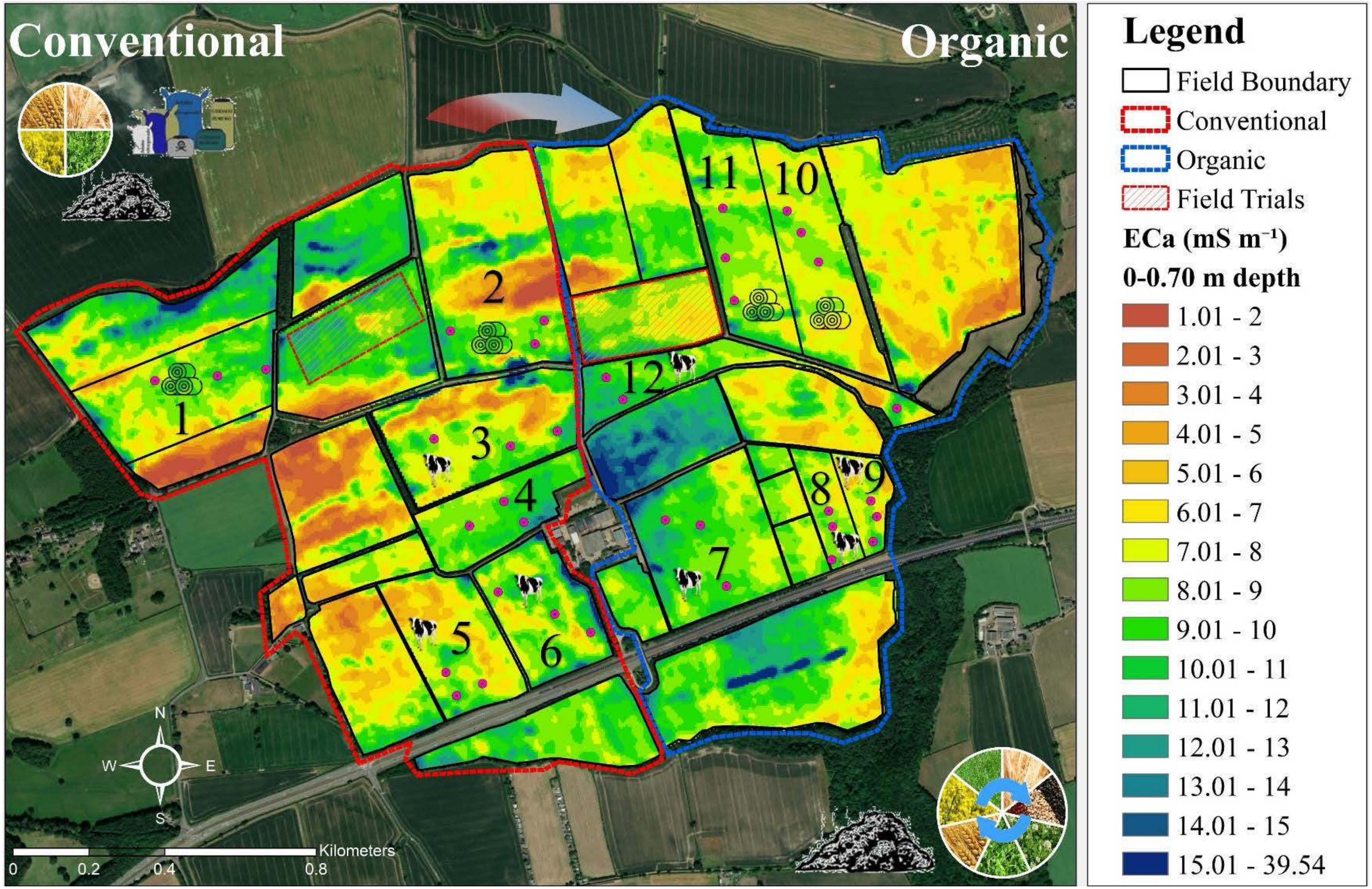
10	organic	Non-grazed	30	70	Ley-arable rotation of wheat, barley and beans in which the field is for the last five years in a row, seven tillage occurrences, and with ley before that for three years in a row with two previous arable rotation. Annual fertilisation (only organic forms) of roughly 67, 74 and 200 kg ha^{-1} yr^{-1} for N, P and K, respectively.
11	organic	Non-grazed	30	60	Ley-arable rotation of wheat, barley and beans in which the field is for the last six years in a row, five tillage occurrences. Before that, ley was used for three years in a row with one previous year under arable. Annual fertilisation (only organic forms) of roughly 71, 79 and 200 kg ha^{-1} yr^{-1} for N, P and K, respectively.
12	organic	Grazed	70	40	Ley-arable rotation of wheat, barley, beans, and ley in which the field is for six years in a row before the three years of arable crops, three tillage occurrences. Annual fertilisation (only organic forms) of roughly 65, 46 and 96 kg ha^{-1} yr^{-1} for N, P and K, respectively.

Table 2. Effects of agricultural system (S) (conventional – CONV and organic – ORG), grazing regime (G) (non-grazed – NG and grazed – GG) and their interaction on soil quality indicators: active acidity (pH), Olsen's phosphorus (P), extractable potassium (K), bulk density (BD), aggregate stability (AS), soil C concentration (C) and microbial biomass carbon (MBC) at three soil depth intervals. Data are measured mean values (n=18 for each S, n=24 for grazed and n=12 for non-grazed). The standard error of the mean in parentheses. Significance tests using likelihood ratio test (LRT), are compared models with or without the parameter of interest. Significant effects ($p < 0.05$) are shown in bold.

		Chemical indicators			Physical indicators		Biological indicators	
Depth (m)		pH	P	K	BD	AS	C	MBC
		H_2O	----------- mg kg^{-1}	-----------	Mg m^{-3}	%	g kg^{-1}	mg kg^{-1}
0-0.15	CONV	6.22 (0.09)	29.42 (2.99)	183.08 (37.12)	1.09 (0.02)	73.62 (2.84)	27.68 (1.11)	181.56 (18.33)
	ORG	6.36 (0.08)	12.25 (2.54)	226.25 (55.32)	1.08 (0.02)	69.16 (2.70)	25.72 (0.92)	236.52 (16.34)
	NG	6.36 (0.08)	13.97 (3.46)	135.74 (35.44)	1.12 (0.03)	65.31 (3.44)	23.24 (0.59)	170.37 (22.59)
	GG	6.25 (0.08)	24.27 (2.99)	239.13 (45.11)	1.06 (0.02)	74.43 (2.19)	28.43 (0.86)	228.37 (14.58)
	S	LRT=0.87; p=0.35	LRT=10.5; **p<0.01**	LRT=0.11; p=0.92	LRT=0.06; p=0.81	LRT=0.95; p=0.33	LRT=1.63; p=0.20	LRT=4.23; **p=0.04**
	G	LRT=0.49; p=0.48	LRT=5.18; **p=0.02**	LRT=1.95; p=0.16	LRT=1.77; p=0.18	LRT=2.86; p=0.09	LRT=9.10; **p<0.01**	LRT=4.19; **p=0.04**
	S*G	LRT=1.44; p=0.23	LRT=0.99; p=0.31	LRT=4.25; **p=0.04**	LRT=5.66; **p=0.02**	LRT=0.02; p=0.88	LRT=1.38; p=0.24	LRT=0.57; p=0.45
0.15-0.30	CONV	6.59 (0.12)	8.72 (0.50)	83.94 (8.03)	1.21 (0.07)	-	20.22 (1.21)	-
	ORG	6.66 (0.10)	9.61 (0.99)	88.44 (15.02)	1.19 (0.07)	-	19.67 (0.59)	-
	NG	6.78 (0.09)	11.00 (1.11)	120.00 (16.36)	1.20 (0.02)	-	18.78 (0.84)	-

	GG	6.54 (0.10)	8.25 (0.54)	69.29 (7.75)	1.20 (0.01)	-	20.53 (0.89)	-
	S	LRT=0.20; p=0.65	LRT=0.21; p=0.64	LRT=0.38; p=0.53	LRT=0.89; p=0.34	-	LRT=0.01 p=0.92	-
	G	LRT=2.17; p=0.14	LRT=3.76; p=0.05	LRT=10.3; **p<0.01**	LRT=0.00; p=0.97	-	LRT=1.60; p=0.23	-
	S*G	LRT=0.65; p=0.42	LRT=2.72; p=0.10	LRT=0.46; p=0.50	LRT=0.36; p=0.55	-	LRT=4.89; **p=0.03**	-
0.30-0.60	CONV	7.12 (0.09)	1.39 (0.14)	58.33 (2.62)	1.29 (0.01)	-	13.20 (1.17)	-
	ORG	7.09 (0.07)	1.78 (0.17)	49.72 (2.61)	1.24 (0.02)	-	10.18 (0.59)	-
	NG	7.14 (0.12)	1.58 (0.23)	54.83 (1.86)	1.24 (0.02)	-	11.88 (1.29)	-
	GG	7.08 (0.06)	1.58 (0.13)	53.63 (2.82)	1.28 (0.01)	-	11.60 (0.84)	-
	S	LRT=0.04; p=0.83	LRT=2.99; p=0.08	LRT=5.00; **p=0.02**	LRT=2.68; p=0.10	-	LRT=6.48; **p=0.01**	-
	G	LRT=0.17; p=0.68	LRT=0.00; p=1.00	LRT=0.10; p=0.75	LRT=1.63; p=0.20	-	LRT=0.01; p=0.91	-
	S*G	LRT=0.70; p=0.40	LRT=2.14; p=0.14	LRT=0.20; p=0.65	LRT=4.04; **p=0.04**	-	LRT=0.50; p=0.47	-

Figure 1. Map of spatial variability of apparent soil electrical conductivity (ECa) 0-0.70 m depth at Nafferton farm showing the 36 locations (pink



points) where the soil cores were taken. Numbers from 1 to 12 refer to the study fields selected across the farm (1-6 conventional and 7-12 organic). Non-grazed and grazed study sites are denoted by hay bales or a cow, respectively.

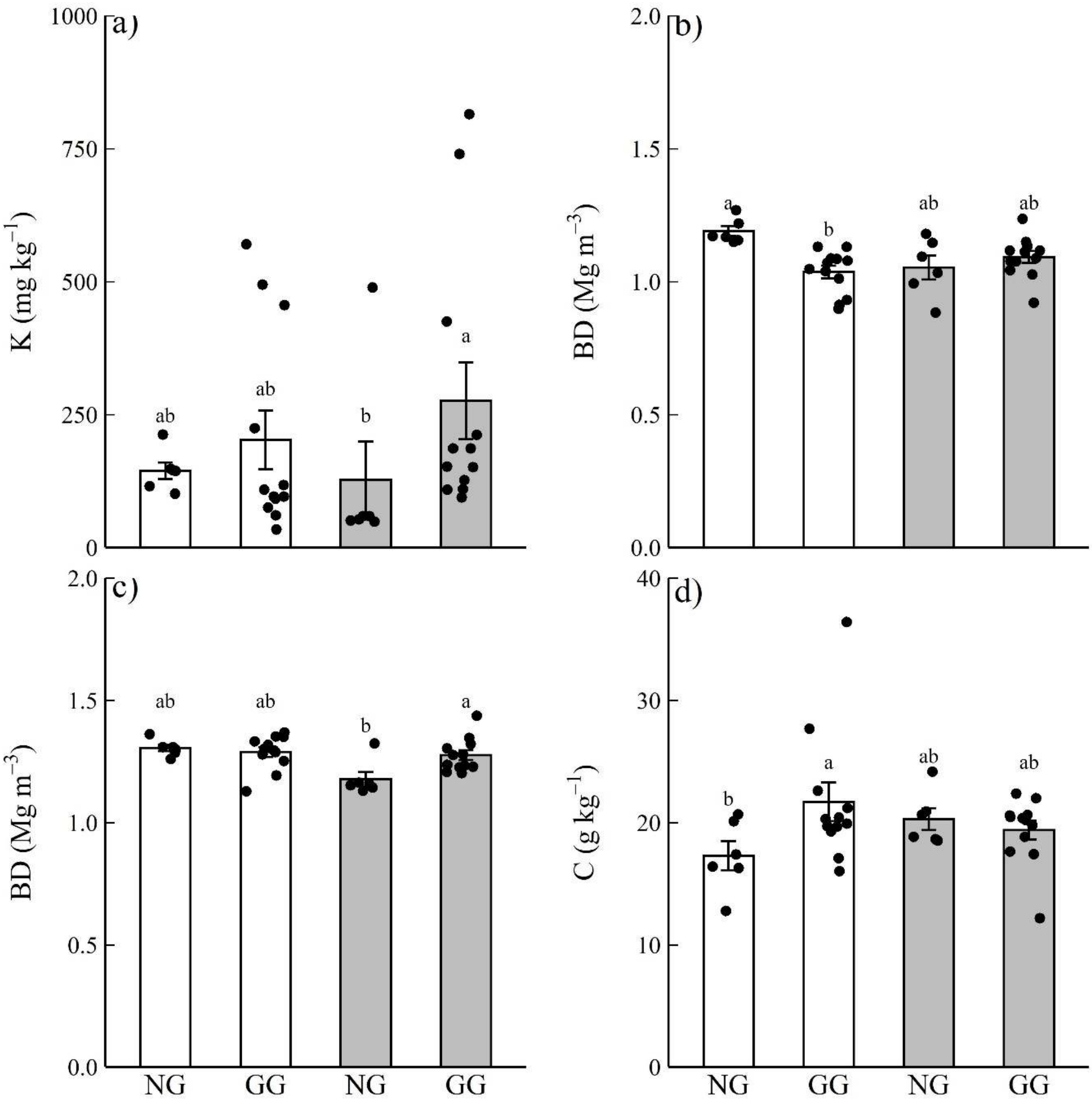


Figure 2. Interactive effects between agricultural system (conventional – CONV and organic – ORG) and grazing regime (non-grazed – NG and grazed – GG) on the following soil quality indicators and soil depth intervals: a) extractable potassium (K) for 0-0.15 m; b) bulk density (BD) for 0-0.15 m, c) bulk density (BD) for 0.30-0.60 m and d) soil carbon concentration (C) for 0.15-0.30 m. Data are measured mean values ± SE (black dots represent individual sample values, n=12 for conventional and organic grazed and n=6 for conventional and organic non-grazed). Significance tests using likelihood ratio test (LRT) comparing models with or without parameter of interest. Mean measured indicator values followed by the same letter do not significantly differ according to Tukey's test ($p < 0.05$).

Figure 3. Relationship between soil quality indicators: active acidity (pH), Olsen's phosphorus (P), extractable potassium (K), bulk density (BD), aggregate stability (AS), microbial biomass carbon (MBC) and soil carbon concentration (C), and ley time proportion (years). Data are measured indicator values (n=36 for each indicator in each soil depth interval 0-0.15, 0.15-0.30 and 0.30-0.60 m). Significance tests using a linear mixed effect model (LME). Significant

effect ($p < 0.05$) is shown in the specific soil indicator figure by depth: blue (0-0.15 m), red (0.15-0.30 m) and black (0.30-0.60 m).