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Indicators of soil ecosystem services in conventional and organic arable fields along a gradient of landscape heterogeneity in southern Sweden

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ABSTRACT

Agricultural intensification has been vital for meeting global food demand but has caused environmental degradation. This has disrupted the ability of soil to provide vital ecosystem services. Organic farming is often thought to conserve and utilise soil ecosystem services, and thus be a more sustainable method of food production than conventional farming. However, evidence for this is equivocal, and little is known of the potential trade-offs between soil functions, which can be classified as supporting and provisioning ecosystem services, in conventional and organic systems. In addition, few studies have simultaneously examined how surrounding landscape heterogeneity affects soil functions in agriculture. In this study we investigated the effects of farming method (conventional versus organic) and landscape heterogeneity (100 m, 500 m and 1000 m radius) on indicators of soil ecosystem services: soil organic carbon (SOC), total nitrogen (TN), water holding capacity (WHC) and plant-available phosphorous (P) (measures of carbon storage and nutrient retention); net N mineralisation and microbial community composition and biomass (nutrient cycling); and crop yield. We found no effect of landscape heterogeneity, and no differences in any of the measured soil and microbial variables between conventional and organic farms, apart from net N mineralisation, which was higher in organic farms. However, conventional farms had significantly greater yield than organic farms, and there was no apparent trade-off between increasing yield and the level of supporting ecosystem services. The organic farms in this study appear to have been intensively managed, with a straight substitution of organic inputs for chemicals, but little other effort to enhance soil fertility. For example, the organic farms applied large quantities of manure compared with conventional farms but conducted mechanical weeding (harrowing), whereas conventional farms applied herbicides. This repeated soil disturbance may cause rapid organic matter mineralisation and undermine the ability of these organic farms to retain carbon and nitrogen. The terms ‘organic’ and ‘conventional’ agriculture both cover a wide variety of farming methods, some of which enhance or deplete ecosystem services more than others. To develop truly sustainable methods of agriculture, research should focus on the effects of specific farming practices, rather than the labels ‘conventional’ and ‘organic’.

1. Introduction

Agricultural intensification has been vital for meeting global food demand with increasing human population growth. But this success has come at a cost: agricultural intensification has led to the degradation of ecosystem services, through increased greenhouse gas emissions, nutrient run-off and biodiversity loss (Bullock et al., 2011; Godfray et al., 2010). Pressure on agricultural production will increase further as the world’s population continues to grow (Foley et al., 2011). The need to develop farming methods that both conserve and utilise ecosystem services for sustainable crop production is therefore of great importance.

Soils provide a range of functions that can be classified into two broad categories: supporting and provisioning services. Supporting services are defined as “those [services] that are necessary for the production of all other ecosystem services”, while provisioning services are defined as “the products obtained from ecosystems” (Millenium Ecosystem Assessment, 2005). Nutrient cycling, which includes carbon storage and nutrient retention, is classed as a supporting service, while food production is classed as a provisioning service (Millenium Ecosystem Assessment, 2005).

Carbon storage and nutrient retention are fundamentally important to agriculture because of their role in maintaining soil fertility through nutrient cycling (Zhang et al., 2007). Soil organic carbon (SOC), the main component of soil organic matter, is the primary resource for the soil microbial community (Bardgett, 2005). Soil microbes decompose organic matter and thereby release nutrients for plant uptake; and the simultaneous conversion of nutrients

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into microbial biomass also reduces the loss of nutrients from the system (Brussaard et al., 1997; Zhang et al., 2007). Therefore, maintaining high levels of SOC is vital for enhancing microbial biomass, which is vital for nutrient cycling, nutrient retention and soil fertility, and as a consequence, crop productivity (Power, 2010). Given these strong linkages, levels of SOC, total nitrogen (TN), phosphorous (P) and microbial biomass can be used as indicators of soil ecosystem services (Bockstaller et al., 1997; Dale and Polasky, 2007).

Conventional agriculture, which typically involves annual crop rotations sustained by repeated application of inorganic N and P fertilisers, has been found to decrease levels of SOC and TN, and reduce soil microbial biomass and diversity. In contrast, organic agriculture, which proscribes the use of inorganic fertilisers and synthetic pesticides, has been found to increase levels of topsoil SOC and TN, and foster greater microbial biomass and diversity (Birkhofer et al., 2008; Mäder et al., 2002; Reganold et al., 2010; Verbruggen et al., 2010). Consequently, organic farming is often upheld as a more sustainable farming method that conserves and utilises supporting ecosystem services (Azadi et al., 2011; Pimentel et al., 2005).

However, there is conflicting evidence on the ability of organic agriculture to improve indicators of soil ecosystem services above that of conventional agriculture. Gosling and Shepherd (2005) found no differences in levels of soil organic matter, TN or C:N ratio between conventionally and organically managed farms. Kirchmann et al. (2007) found that organic farms performed no better than conventional farms across a range of measures, including nitrogen leaching and loss of SOC, and produced lower yields. Yields from organic farms are often less than conventional farms (Birkhofer et al., 2008; Mäder et al., 2002; Noponen et al., 2012), though this is not always the case (Seufert et al., 2012).

One reason for the equivocal results from comparisons of conventional and organic systems may be that the definitions of what can be certified as 'organic' tend mainly to regulate inputs, i.e. pesticides and fertilisers. This limited definition may be necessary for clarity of certification standards, but can potentially ignore a range of farming methods that different organic farmers might implement (Rigby and Cáceres, 2001). In other words, there are likely to exist intensive and extensive organic farms, and depending on which have been selected for comparison with conventional farms, differences in soil services may or may not be apparent. The same is true in reverse for conventional farms, which can include farms managed in an integrated way or with reduced tillage, but are classed as 'conventional' due to inorganic inputs (Trewavas, 2004).

In addition to comparing indicators of individual ecosystem services, it is important to analyse trade-offs between services. In a landscape-scale spatial analysis, Raudsepp-Hearne et al. (2010) found a strong trade-off between the value of provisioning services (food and timber) and supporting services (carbon sequestration and nutrient cycling). In order to identify management that delivers multiple ecosystem services as opposed to one at the expense of others, it is important to understand the trade-offs between ecosystem services (Bennett et al., 2009), and in this case within conventional and organic agriculture.

In addition to field scale factors, landscape factors can also affect ecosystem services in agricultural fields. Heterogeneous landscapes (those containing a mixture of agricultural fields, permanent pastures and woodlands) have, in general, been found to support greater biodiversity than homogenous landscapes (those that are predominantly agricultural fields), irrespective of farming intensity (Rundlöf and Smith, 2006; Tscharntke et al., 2005). However, this relationship has been demonstrated primarily for above-ground organisms (Blitzer et al., 2012), with soil organisms largely overlooked, possibly because they are relatively immobile (Bardgett, 2005). However, one recent study found that soil microbial biomass was dependent on both the farming system and landscape heterogeneity: microbial biomass was enhanced in conventional fields within heterogeneous landscapes but reduced within homogenous landscapes. Conversely, in organic fields microbial biomass was enhanced in homogenous landscapes and reduced in heterogeneous landscapes (Flohre et al., 2011). This result was hypothesised to have arisen out of the potential for heterogeneous landscapes to support greater numbers of aboveground predators that would have cascading effects on the belowground community (Flohre et al., 2011). This demonstrates the importance of considering landscape heterogeneity when comparing soil ecosystem services within conventional and organic farming systems.

The aim of this study was to elucidate how conventional and organic farming affect the delivery and trade-off of soil ecosystem services, and whether this is affected by landscape context. We did this by measuring and comparing a range of indicators of soil ecosystem services within conventional and organic arable fields across a landscape heterogeneity gradient in southern Sweden. Specifically, we investigated levels of SOC, TN, plant-available P and water holding capacity (WHC) (measures of carbon storage and nutrient retention); net N mineralisation rate and microbial community composition and biomass (nutrient cycling); and crop yield (production). We then examined the relationship between these soil variables to identify any trade-offs between provisioning and supporting ecosystem services. Questionnaires were also sent to each of the farmers to provide information on fertiliser use, weed control, soil preparation and crop rotations.

2. Methods

2.1. Site selection and sampling

Farms in Scania, southern Sweden, were selected for sampling based on their surrounding landscape heterogeneity. This was calculated using scripts developed in MATLAB 7.11.0 on data from the Swedish Board of Agriculture's Integrated Administrative and Control System database (IACS, Blockdatabasen). Landscape heterogeneity was described by the combination of the amount of permanent pasture and field border within 1000 m radius landscapes centred over each farm. The length of field borders was translated into an area by assuming field borders have a width of 1 m. The amount of permanent pasture and field border was then expressed as proportions of the total agricultural area within landscapes. The proportions of permanent pasture and field border were combined by extracting the first principal component (PC1) from a principal component analysis (PCA) of the two variables (prior to analysis proportion of pasture was square root transformed to improve linearity). To ensure that landscapes varied more along PC1 than PC2, all landscapes that had a standard deviation along PC2 greater than one were excluded. All landscapes that contained less than 40% farmland were also excluded. All remaining conventional and organic farms were then plotted against PC1 so that as large variation in landscape heterogeneity as possible could be sampled (higher values of PC1 indicate more heterogeneous landscapes; lower values indicate homogenous landscapes). A total of 17 farms were selected for sampling: 10 conventional and 7 organic. All the organic farms had been organically managed for at least 10 years. The area of permanent pasture within these landscapes ranged from 0 to 56 ha; while the area of field border ranged from 2 to 6 ha. After site selection, two additional PC1 values were calculated for each farm using 100 m and 500 m radius landscapes. The three PC1 values per farm were then used to investigate the effect of landscape heterogeneity at three spatial scales: 100 m, 500 m and 1000 m (PC_{100}, PC_{500} and PC_{1000}, respectively).

Soil samples were collected from fields of spring barley (*Hordeum vulgare* L.) from the selected farms during May–June 2011. The soils of the study area are predominantly Eutric Cambisols with pockets of Dystric Cambisols. Four samples were taken from each field and bulked to provide a representative sample of that field. Where available four samples were also taken from adjacent permanent pastures owned by the same farmer (seven pastures in total), to provide a low-intensity land use comparison. Samples were taken from the top 20 cm of soil using a 3 cm diameter soil corer. Samples were kept in sealed plastic bags within cool boxes until reaching the laboratory, where they were bulked and sieved to 2.5 mm. Each bulked sample was then split into two sub-samples, one of which was frozen at −20 °C and the other refrigerated at 4 °C. The geographic coordinates of all sampled farms are given in Appendix A.

The farmers' responses to the questionnaire showed that all of the barley fields had undergone soil preparation prior to seed sowing (ploughing or tillage). All of the conventional farmers applied either inorganic fertilisers or a mix of inorganic fertilisers and manure; the organic farmers applied only manure. To compare the amount of immediately available N (avN) in manure with inorganic fertilisers, it was assumed that manure contained 1.2 kg avN per tonne of manure (DEFRA, 2010). Average avN from fertiliser application was 160 kg avN ha^{-1} in conventional fields, and 26 kg avN ha^{-1} in organic fields. All the conventional farmers treated their fields with various chemical herbicides, whereas the organic farmers only conducted mechanical weeding. Five year crop rotations differed between conventional and organic fields: conventional fields underwent annual monocropping with rotations based on barley, wheat (*Triticum aestivum* L.) and sugar beets (*Beta vulgaris* L.); organic fields underwent less intense cropping, including up to 3 years under ley (hay production). The pastures were grazed by either cattle or sheep, and were not ploughed, fertilised or treated with pesticides.

2.2. Laboratory analyses

Three sub-samples were taken from each of the frozen samples. The first set of sub-samples was weighed, oven-dried at 100 °C and reweighed to determine gravimetric water content. These were then soaked with water for 48 h to reach field capacity and reweighed for WHC. The second set of sub-samples was oven-dried at 60 °C before being analysed for pH (1:2.5, w/v H_2O), SOC (CO_2 evolution after ignition at 1050 °C with cobalt oxide catalyst), TN (Kjeldahl digest), and plant-available P (Bray-1 extraction; Bray and Kurtz, 1945). The refrigerated samples were used for determination of net N mineralisation; each sample was adjusted to 60% WHC and incubated at 25 °C for 21 days. Total ammonium and nitrate (2 M KCl extraction) were measured before and after incubation.

The third set of frozen sub-samples was used for fatty acid analysis of the soil microbial community. Lipids were extracted from 3 g (fresh weight) of soil in a one-phase mixture of citrate buffer, methanol and chloroform (0.8:2:1, v/v/v, pH 4.0) (Bligh and Dyer, 1959; White et al., 1979). The lipids were then fractionated into neutral lipids, glycolipids and phospholipids using pre-packed silica columns (100 mg sorbent mass, Varian Medical Systems, Palo Alto, USA), by elution with chloroform, acetone and methanol, respectively (Frostegård et al., 1991). The lipids were transformed to fatty acid methyl esters by mild alkaline methanolysis, and analysed by gas chromatography using a 50 m HP 5 capillary fused silica column (Hewlett Packard, Palo Alto, USA), using H_2 as carrier gas (Frostegård et al., 1993).

Each fatty acid was identified from their retention time relative to that that of the internal standard (fatty acid methyl ester 19:0). Phospholipid fatty acids (PLFAs) i15:0, a15:0, 15:0, i16:0, 16:1ω7, 16:1ω9, i17:0, a17:0, cy17:0, 17:0, 18:1ω7 and cy19:0 were used to represent bacterial biomass; 18:2ω6 was used to represent biomass of saprotrophic fungi (Frostegård and Bååth, 1996). PLFAs 10Me16:0, 10Me17:0 and 10Me18:0 were used to represent actinomycetes (Aliasgharzad et al., 2010). The neutral lipid fatty acid (NLFA) 16:1ω5 was used to represent biomass of arbuscular mycorrhizal fungi (AMF) (Olsson, 1999). PLFA and NLFA nmol concentrations were converted to biomass C using the following factors: bacterial PLFAs: 363.6 nmol = 1 mg C; fungal PLFA: 11.8 nmol = 1 mg C; and AMF NLFA: 1.047 nmol = 1 μg C (Frostegård and Bååth, 1996; Klamer and Bååth, 2004; Olsson et al., 1995).

2.3. Statistical analyses

Soil chemical and physical properties was analysed separately using contrasts (conventional versus organic; and conventional + organic, henceforth termed 'arable fields', versus pasture) with linear mixed effects models fitted with restricted maximum likelihood estimations (Zuur et al., 2009). The fixed effects were management type (the contrasts), soil pH, the three PC1 values (PC_{100}, PC_{500} and PC_{1000}), and their interactions. Scania is blocked into yield regions, and these were fitted as a random effect. Relationships between individual soil properties were analysed using Pearson's product moment correlations. Bacterial and fungal biomass were analysed similarly, including Bray P as a fixed effect. Barley yield was also analysed with a mixed effects model, using the conventional versus organic contrast and yield region as random blocking factor. Variables were $\log_e$-transformed where necessary. Trade-offs between ecosystem services were analysed by nonmetric multidimensional scaling (NMDS) using Bray–Curtis distances. To account for natural variation in SOC from different soil types, the value of SOC in the barley fields was expressed as the ratio of barley field SOC to the mean SOC of the seven permanent pastures. The permanent pastures represent SOC under low intensity soil management, thus a ratio close to one indicates soil with high SOC, a ratio close to zero indicates low SOC.

Microbial community composition was analysed by distance-based redundancy analysis (Legendre and Anderson, 1999) and PERMANOVA: a Bray–Curtis distance matrix of PLFA nmol concentrations was analysed against the management types and environmental variables (soil chemical properties and the three PC1 values). All analyses and graphical operations were conducted in R 2.14.2 (R Development Core Team, 2012), using packages nlme (Pinheiro et al., 2012) and vegan (Oksanen et al., 2011).

3. Results

3.1. Soil properties, barley yield and trade-offs

SOC, TN, Bray P and WHC did not differ between conventional and organic barley fields, but did between arable fields (conventional + organic) and pastures; pastures had significantly higher levels of all variables except Bray P (Table 1). Organic barley fields had a significantly higher rate of net N mineralisation than conventional barley fields ($p = 0.044$). Net N mineralisation rates of arable fields and pastures did not differ significantly (Table 1). TN and WHC had a significant linear relationship with SOC (TN: $t_{1,22} = 10.19$, $p < 0.001$, $r^2 = 0.83$; WHC: $t_{1,22} = 5.92$, $p < 0.001$, $r^2 = 0.61$). Landscape heterogeneity at all spatial scales (PC_{100}, PC_{500} and PC_{1000}), soil pH and their interactions had no effect on any of the soil properties.

Barley yield was significantly different between conventional and organic barley fields ($t_{1,9} = 4.19$, $p = 0.002$). Grain yield from the conventional farms was 6.0 ± 0.36 tonnes ha^{-1}; yield from the organic farms was 4.0 ± 0.20 tonnes ha^{-1}. For the NMDS analysis,

Table 1
Soil chemical and physical properties (mean ± 1S.E.). Different letters within rows and contrasts indicate significant differences.

Soil properties	Conventional versus organic			Arable (conventional + organic) versus pasture		
	Conventional	Organic	Test statistic	Arable	Pasture	Test statistic
SOC ($g\,kg^{-1}$)	9.82 ± 1.30^a	9.73 ± 0.94^a	$t_{1,9} = 0.06$	9.78 ± 0.84^a	16.41 ± 1.28^b	$t_{1,16} = 4.30$
TN ($g\,kg^{-1}$)	1.72 ± 0.13^a	1.67 ± 0.12^a	$t_{1,9} = 0.27$	1.70 ± 0.09^a	2.37 ± 0.08^b	$t_{1,16} = 4.60$
WHC (%)	53 ± 2^a	49 ± 2^a	$t_{1,9} = 1.18$	51 ± 2^a	77 ± 6^b	$t_{1,15} = 7.07$
Bray P ($mg\,kg^{-1}$)	154.82 ± 38.68^a	191.34 ± 51.81^a	$t_{1,9} = 0.59$	169.86 ± 30.49^a	115.57 ± 39.24^a	$t_{1,16} = 1.45$
Net N mineralisation ($mg\,kg^{-1}$)	14.56 ± 1.07^a	16.09 ± 2.34^b	$t_{1,8} = 2.39$	15.23 ± 1.16^a	23.59 ± 6.89^a	$t_{1,15} = 0.69$
pH	5.6 ± 0.2^a	5.3 ± 0.1^a	$t_{1,9} = 0.94$	5.5 ± 0.1^a	5.6 ± 0.2^a	$t_{1,16} = 0.39$

SOC was taken as a surrogate for TN and WHC. Two conventional and two organic farmers did not provide fertiliser application data, thus a reduced dataset was used for this analysis. No trade-offs between ecosystem service indicators were found, i.e. increasing barley yield had no effect on SOC, TN, WHC, net N mineralisation or Bray P. PERMANOVA showed that the only differences between conventional and organic fields were yield and fertiliser avN ($F_{1,11} = 4.69$, $p = 0.003$; Fig. 1).

3.2. Soil microbial community

Bacterial and fungal biomass C did not differ significantly between conventional and organic barley fields, but did between arable fields and pastures, where pastures had significantly greater biomass (Table 2). Fungal biomass C had a positive relationship with soil pH, but was not significant ($p = 0.060$). Biomass C of AMF did not differ between conventional and organic fields, but did between arable fields and pastures; AMF biomass in pastures was approximately three times greater than that in arable fields (Table 2). Biomass C of AMF also showed a negative relationship with plant-available P ($t_{1,22} = 3.01$, $p = 0.006$; Fig. 2).

Analysis of the PLFA signatures showed that conventional and organic barley fields did not differ in microbial community composition ($F_{1,15} = 0.14$, $p = 0.89$), but together differed from pastures ($F_{1,21} = 18.92$, $p < 0.001$; Fig. 3). The only significant soil term in the analysis was SOC ($F_{1,21} = 17.10$, $p < 0.001$). The abundance of all PLFAs increased with SOC. Actinomycetes appeared most abundant with decreasing SOC and increasing P, i.e. within the barley fields. Landscape heterogeneity at all spatial scales and plant-available P had no significant effect on soil microbial community composition.

4. Discussion

In this study, and for the measured indicators, supporting soil ecosystem services do not differ between conventionally and organically managed arable fields. The organic fields received a higher annual dosage of organic material, in the form of manure, and underwent slightly less intensive cropping by including leys in their rotations (the conventional farmers all conducted annual cropping). Despite this, levels of SOC, TN and WHC were no different to that of the conventional fields. These results therefore align with those of Gosling and Shepherd (2005), Kirchmann et al. (2007) and Bell et al. (2012), and fall contrary to those who have found higher levels of ecosystem services in organically managed soils (Birkhofer et al., 2008; Mäder et al., 2002; Reganold et al., 2010).

One reason for the absence of differences in soil properties may be related to weed management. Farmer responses to the management practices questionnaire showed that all conventional farmers used herbicides for weed control while organic farmers weeded mechanically, by harrowing. Frequent disturbance of the upper soil layers increases organic matter mineralisation rates and decreases levels of topsoil C and N (Balesdent et al., 2000; Lal and Kimble, 1997). Soil organic matter in topsoil is important for improving soil structural properties as well as water infiltration and retention rates (Blanco-Canqui and Lal, 2008; Endale et al., 2010). Organic matter is also the primary substrate sustaining the soil microbial community and thus nutrient cycling (Bardgett, 2005). In addition, mechanical weeding does not control weeds as effectively as herbicides and can result in reduced crop yields (Ryan et al., 2010; Teasdale et al., 2007; Tørresen et al., 2003). Therefore, the soil ecosystem services that could potentially be accrued with use of manure fertilisers (increased carbon storage and nutrient

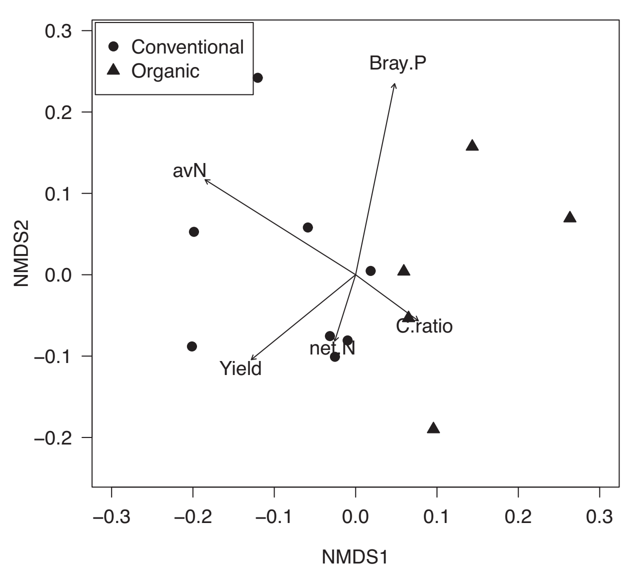

Fig. 1. NMDS of soil services. NMDS1 separates the sites according to yield and fertiliser available N (avN); NMDS2 separates the sites according to Bray P.

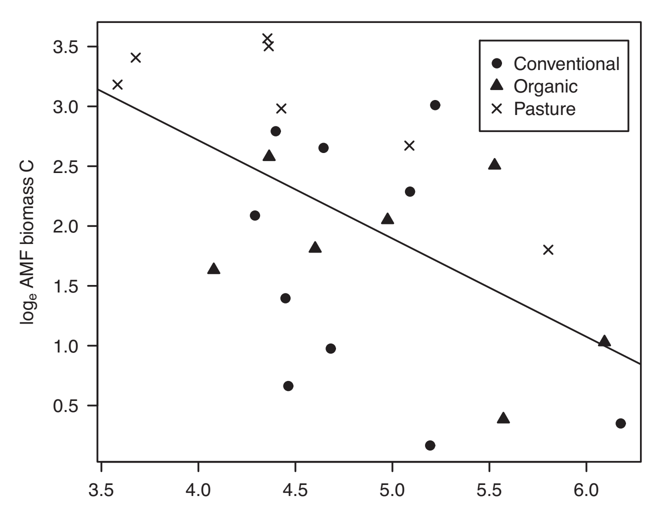

Fig. 2. AMF biomass C and plant-available P (Bray-1 extraction). $\ln y = -0.62(\ln x) + 6.0$; $r^2 = 0.29$.

Table 2
Bacterial, fungal and AMF biomass C (mean ± 1S.E.). Different letters within rows and contrasts indicate significant differences.

Microbial biomass	Conventional versus organic			Arable (conventional + organic) versus pasture		
	Conventional	Organic	Test statistic	Arable	Pasture	Test statistic
Bacteria (mg C g^{-1})	0.21 ± 0.02[a]	0.20 ± 0.01[a]	$t_{1,9}$ = 0.57	0.21 ± 0.02[a]	0.34 ± 0.02[b]	$t_{1,16}$ = 4.02
Fungi (mg C g^{-1})	0.32 ± 0.07[a]	0.22 ± 0.02[a]	$t_{1,9}$ = 1.12	0.28 ± 0.04[a]	0.44 ± 0.10[b]	$t_{1,16}$ = 2.16
AMF (μg C g^{-1})	8.0 ± 2.19[a]	6.96 ± 1.68[a]	$t_{1,9}$ = 0.35	7.57 ± 1.43[a]	23.31 ± 4.03[b]	$t_{1,15}$ = 2.92

retention and cycling), may not be being realised on these organic farms due to inefficient weeding practices (Trewavas, 2001).

The only measured soil variable to differ between conventional and organic fields was net N mineralisation rate. This was found to be significantly higher in organic fields. Given that microbial community composition and biomass did not differ between conventional and organic fields, it is not clear what caused this difference. It is possible the increased soil disturbance in organic fields, as already discussed, stimulated greater N mineralisation rates. This suggests a possible ecosystem disservice in the organic fields. Fungal biomass is important for N retention (de Vries et al., 2011), and was the same in conventional and organic fields. This poses the risk that greater N mineralisation in the organic fields could result in greater N loss, causing leaching into surrounding habitats and reducing soil fertility. However, given that organic farming systems tend to N limited (Berry et al., 2002), mineralised N may be rapidly taken up by the crop plants and thus not pose a leaching risk.

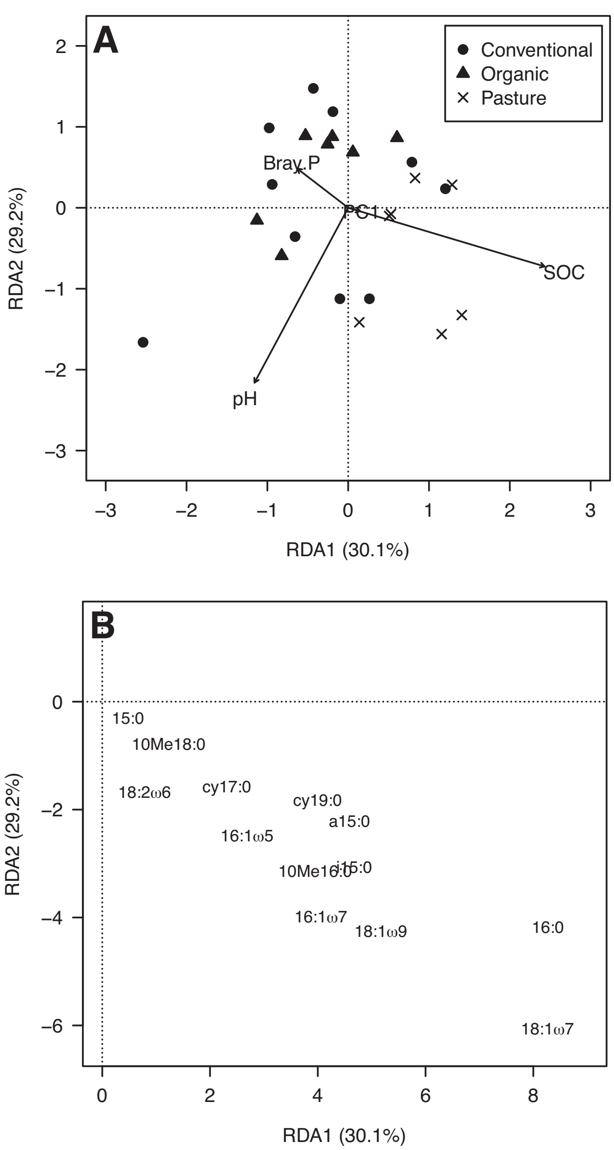

Fig. 3. Distance-based RDA of the soil microbial community. Total explained variation of each axis is shown in parenthesis. (a) RDA scores for each of the sampled fields and the environmental vectors. (b) PLFA loadings for all soil samples.

We expected supporting soil services to decrease with increasing yield as a consequence of intensive soil management (Raudsepp-Hearne et al., 2010). However, no trade-offs were found, irrespective of farming system. The conventional and organic fields in this study were distinguished by only three factors: yield, fertiliser derived available N (both greater in the conventional fields), and weed management (harrowing versus herbicides). In terms of management, therefore, the conventional and organic fields in this study were similar: both were managed intensively. This is particularly clear when the soil properties are compared against the pastures, which were managed in a non-intensive way. Management of the organic fields appears to have attempted a straight substitution of organic inputs for conventional ones. The outcome has been inefficient weeding practices that simultaneously increase SOC mineralisation, and reduced N availability; both of which lowered yields.

For the organic farms in this study to increase levels of available N to those of the conventional fields, using a straight substitution approach, manure application would have to increase six-fold from 20–30 tonnes ha^{-1} to approximately 150 tonnes ha^{-1}. Such an increase might also increase levels of topsoil SOC above those of the conventional fields. However, at such a high application rate, N leaching from manure is likely to be a major problem, especially during autumn and winter (Kirchmann et al., 2002). The quantity of available N applied on the conventional fields was approximately 160 kg N ha^{-1}. Research on cereals in Sweden and the UK suggests that this rate of application will result in leaching of between 20 and 50 mg NO_3—N L^{-1} yr^{-1}, which could be reduced by using catch crops (Goulding et al., 2000; Torstensson and Aronsson, 2000). The EU limit for NO_3-N concentration in groundwater is 50 mg NO_3—N L^{-1} (EC, 2006).

Biomass of AMF hyphae showed a decreasing trend with increasing plant-available P. This relationship has been found elsewhere (Liu et al., 2012), and shows that excessive P fertilisation can decrease the abundance of AMF. Plant-available P and AMF biomass were statistically equivalent in conventional and organic fields, indicating that the sampled organic farms are no more conducive to supporting/diminishing AMF abundance than the conventional farms. However, organic agriculture has been found to support greater AMF diversity than conventional agriculture (Verbruggen et al., 2010). A recent glasshouse study also found that sterilised soil inoculated with AMF from organic fields developed greater AMF hyphae than inoculum from conventional fields, and that P leaching from soil after simulated rainfall decreased with more AMF hyphae (Verbruggen et al., 2012). However, the same study also found that colonisation of plants by AMF from organic fields resulted in plant biomass reductions more often than AMF from conventional fields. Further research is required to disentangle how agricultural practices affect AMF diversity, and how this translates into hyphal biomass and ecosystem services.

Our results found no effect of landscape heterogeneity on soil functions and microbial community composition. This points to a stronger influence of local rather than landscape effects on soil processes. This is contrary to the findings of Flohre et al. (2011), who found that microbial biomass increased in conventional fields relative to organic fields in heterogeneous landscapes, but decreased in conventional fields relative to organic fields in homogenous landscapes. It is not clear why our results differ from theirs, but it may be related to which measure of landscape heterogeneity is used. Flohre et al. (2011) used percent arable land, whereas we used a combined measure of the length of field borders and the proportion of permanent pastures within the landscapes. Furthermore, to ensure that our sampling was focussed on predominantly agricultural landscapes, we only sampled areas that had a minimum of 40% arable land within the 1000 m radius landscapes. This may have been an important factor in our analysis, as it restricted our sampling to more intensively farmed landscapes, where incremental changes in landscape heterogeneity may have had only a minor effect.

The measured indicators of supporting soil ecosystem services were consistently lower in the barley fields than they were in the pastures, highlighting that soil functions are depleted with increasing intensity of soil management. Organic agriculture, with its emphasis on chemical proscription and reduced cropping intensity, is often said to conserve soil ecosystem services and therefore be more sustainable than conventional agriculture (Azadi et al., 2011; Pimentel et al., 2005). In this study we found no evidence to support this view, as soil functions were similar between conventional and organic farms. In addition, yields from organic farms were significantly lower than those from conventional farms and there was no apparent trade-off between yield and supporting ecosystem services. The straight substitution of organic inputs for chemicals, as was the case here, resulted in the organic farms having lower fertiliser derived available N. This translated into lower crop yields with no compensation in terms of increased soil function. Developing sustainable forms of agriculture is essential to meet the dual needs of increasing food production for a growing world population, and to decrease environmental damage caused by agriculture (Foley et al., 2011). To achieve this, a shift in approach to focus on the effects of specific management practices on ecosystem services, rather than on the broad and heterogeneous definitions of ‘conventional’ versus ‘organic’ farming, seems necessary.

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Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.apsoil.2012.12.019.

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