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Actinobacterial community dynamics in long term managed grasslands

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Abstract

Organic management is believed to 'enhance' nutrient cycling and microbial diversity in soil, yet the extent of this enhancement remains largely unknown. Actinobacteria play a major role in organic matter turnover, carbon cycling and suppression of some fungal pathogens. Thus, we investigated the long-term impact of organic and mineral fertilisers on the actinobacterial community structure in grassland soils using terminal restriction length polymorphism (T-RFLP). Multivariate statistics was then used to investigate the relationship between community structure, seasonality and fertiliser management. Fertilisers have had a profound impact on soil pH with the organically fertilised plots showing a significantly higher pH than those receiving mineral fertilisers. Consequently, soil pH was the most significant edaphic factor influencing actinobacterial communities with *Arthrobacter* and *Micrococcus* more abundant in soils receiving organic inputs whilst *Streptomyces*, *Acidimicrobium* and *Actinospica* more prevalent in acid soils. Soil water content was also influential, with major changes evident over the summer months between May and September. Quantitative PCR of the actinobacterial and fungal 16S and 18S rRNA genes, respectively suggested that fungal gene abundance was negatively correlated ($P = 0.0131$) with increasing actinobacterial gene abundance. The importance of these findings in terms of fungal abundance and potential disease suppression are discussed.

Key Words

Actinobacteria, fertiliser management, 16S rRNA genes, quantitative PCR, disease suppression, C cycling.

Introduction

Actinomycetes are an important and functionally diverse group of organisms that are known to be involved in carbon cycling and nutrient transformation in soils (Goodfellow and Williams 1983). This role in sustaining soil processes reflects their metabolic diversity, their ability to produce secondary metabolites and their mycelial growth habit which, like soil fungi, makes it possible to explore the bulk soil in search of water and nutrients (McCarthy and Williams 1990). There are increasing concerns over the impacts of climate and land use change on the maintenance of soil functions (O'Donnell *et al.* 2001) and the need to increase crop production to meet the diverse demands of food, fibre and biofuel production. To meet these demands requires a better understanding of how soils function and ultimately, the ability to engineer diversity: function relationships so as to maintain and even enhance key system processes (O'Donnell *et al.* 2001). The use of organic inputs in the form of farm yard manure (FYM) and composts are considered to be important in increasing microbial diversity and enhancing nutrient cycles in soils (Mader *et al.* 2002). However, the real extent to which organic and inorganic fertilisers 'enhance' the diversity of different functional and taxonomic groups in soils remains largely unexplored.

Long-term field sites such as the Palace Leas Hay Meadow Experiment, Northumberland, UK, are ideal sites on which to study the impacts of extended fertiliser use on the structure and activity of soil microbial communities since they offer the opportunity of separating the long-term effects of management from shorter-term, temporal variability. The site was established in winter 1896-97 since then the 13 plots have received regular and virtually unchanged mineral fertiliser and FYM inputs. In this study, we have used a combination of molecular fingerprinting using terminal restriction fragment length polymorphism (T-RFLP) and 16S rRNA gene cloning to investigate the structure and seasonal dynamics of actinobacterial communities. Our aim was to investigate whether the different fertiliser regimes had impacted on the actinobacterial community structure and to identify using multivariate statistical techniques the relative importance of environmental parameters such as pH, time of sampling, and the form and rate of fertiliser addition in shaping these communities.

Methods

Sampling regime and environmental variables

To differentiate fertiliser effects from seasonal variation, soils were sampled in triplicate from each of the 13 plots three times over one growing season between May and September 2004 and January 2005. Soil pH and water content were also measured which together with data on the fertiliser inputs and sampling time (January, May or September) provided the environmental variables used in the multivariate statistical analyses

Actinobacterial community profiling and cloning

Total nucleic acids were extracted from 0.5 g soil using previously described methods (Griffiths *et al.* 2000). Actinobacterial community composition was profiled using T-RFLP, where individual peaks represent taxon (OTU) richness and the height of the peaks gives an indication of their relative abundance. Community DNA was amplified using actinobacterial specific primers targeting the 16S rRNA gene where the 5' end was labelled with 6-carboxy-fluorescein (FAM). Following amplification PCR product were digested with restriction enzyme HhaI, desalinated and denatured prior to injecting the samples into the MegaBace 1000 DNA Sequencing System. To assess changes in actinobacterial community composition, reference clone libraries were generated for each plot at Palace Leas in order to identify TRF peaks found in the community profiles of all samples. The TRF of each clone was determined experimentally as described for the soil DNA and also checked *in silico* using published procedures (Egert and Friedrich 2003). Cloned 16S rRNA genes were then amplified, labelled, restricted and run on the MegaBACE as described for the soil DNA. The resulting T-RFLP profiles were analyzed using MegaBACE Genetic Profiler Version 1.5 and used to generate a data matrix of relative taxon abundances for each of the actinobacterial communities. A corresponding matrix of the environmental variables (time of sampling, soil pH, soil water content and the rates and forms of fertiliser inputs) for each of the plots was also prepared. To explore which of these parameters best explained the differences in actinobacterial communities between plots, canonical correspondence analysis (CCA) was used to model the changes in the community profile of the different plots relative to the measured environmental variables.

Fungal and actinobacterial abundance using quantitative PCR

Total gene abundances for actinobacterial and fungal communities in each plot were quantified by real-time PCR (qPCR) using actinobacterial specific and fungal specific primer sets. qPCR was performed using a Roche LightCycler instrument and a SYBR Green I fluorophore protocol. The relationship between actinobacterial and fungal population sizes were investigated using actinobacterial abundance as a predictor of fungal abundance by fitting generalised linear models (GLM). Residuals were assessed for normality using quantile plots and all regression analyses were performed using the R statistical package version 2.0.

Results and discussion

In Figure 1a, the highest correlation between environmental variables and the differences in actinobacterial community structure between plots was obtained with soil pH on axis 1. This means that of the measured variables, soil pH was the most important driver of changes in actinobacterial community composition in the plots at Palace Leas and this has been shown in previous studies (O'Donnell *et al.* 2001; Fierer and Jackson 2006). Figure 1b showed that the increase in pH due to the addition of FYM (contains liming properties, such as Ca^{2+} or Mg^{2+}) to plots 1-5 is accompanied by a change in actinobacterial community structure and an increase in the relative abundance of *Micrococcus*, *Norcardia*, *Mycobacterium* and *Arthrobacter* and these taxa are particularly associated with manures and organic compost (Atagana 2004). Figure 1 also showed that the extended use of NH_2SO_4 as a source of inorganic N has had a major impact on the actinobacterial community structure in plots 7 and 11, with an increase in the relative abundance of acidophilic taxa such as *Acidimicrobium*, *Streptomyces* and *Actinospica* (Figure 1b). These results provide good support for the hypothesis that long-term fertilizer inputs and pH in particular have not only changed the community structure but also the taxonomic diversity of actinobacteria in these soils.

As seen in Figure 1, time of sampling was also identified as an important covariate and therefore actinobacterial community structure changes during the year. Since, time of sampling was correlated to soil water content these changes are most likely a consequence of seasonal differences in rainfall that result in changes in soil water content. This suggests that soil water content acts selectively on actinobacterial communities, possibly favouring desiccation tolerant taxa during the drier months, such as, *Arthrobacter* that can exist for long periods as resting cocci to avoid desiccation (Goodfellow and Williams 1983). When the

soil water content was at its highest in January there was a marked increase in the relative abundance of *Mycobacterium*, *Streptomyces* and *Micrococcus* in all plots. Perhaps these taxa are drought intolerant and periods of soil moisture limitation may affect the physiological status of these bacteria through starvation, induced osmotic stress, and resource competition.

Figure 2 shows the variation in fungal and actinobacterial biomass between plots as estimated using qPCR amplification of the 16S rRNA and the 18S rRNA genes respectively. One of the benefits claimed for organic agriculture is that organic inputs help limit the growth of soil fungal pathogens (Bailey and Lazarovits 2003). The data presented here supports this view since fungal abundance was lowest in those plots receiving annual inputs of FYM (plots 1 and 2) but that the effect was lost when FYM was applied in alternate years with NPK fertilizer (plot 3). If one considers what was happening in plots 4 and 5, where FYM was added in alternate years but without NPK (plot 4) and at double the rate followed by 4 years of NPK (plot 5) it appears that these management induced changes in actinobacterial and fungal dynamics are related to both the amounts and timing of FYM inputs. Although the high correlation between the abundance of actinobacteria and fungi was interesting in terms of the potential control of soil-borne fungal diseases, there was yet no direct evidence that there are any differences in disease suppression between treatments or that there was any direct interaction or antagonism between actinomycetes and fungi.

Conclusion

The data presented in this study show clearly that soil pH is a major driver of change in actinobacterial communities. The data also suggests that at the level of individual taxa these responses are consistent with previously published results of isolation studies with genera such as *Arthrobacter* and *Micrococcus* more abundant in soils receiving organic inputs and others such as *Streptomyces*, *Acidimicrobium* and *Actinospica* more prevalent in acid soils. The actinobacteria have also been shown to respond to other environmental drivers such as time (seasonal variation) in a manner consistent with changes in soil water content throughout the year. Again these findings are consistent with published reports of isolation studies and could have an important bearing on the design and optimisation of selective isolation and screening programmes. In addition to these abiotic interactions the T-RFLP and qPCR data have shown that management and pH in particular have impacted on the biological interactions in these soils with elevated actinomycete numbers consistent with a decrease in soil fungal propagules. However, the data also show that this effect can be lost where organic fertiliser inputs are not applied annually as in plots 4 and 5. The reasons for this are unknown and need to be further investigated.

References

- Atagana HI (2004) Co-composting of PAH-contaminated soil with poultry manure. *Letter of Applied Microbiology* **39**, 163-168.
- Bailey KL, Lazarovits G (2003) Suppressing soil-borne diseases with residue management and organic amendments. *Soil Tillage Research* **72**, 169-180.
- Egert M, Friedrich MW (2003) Formation of pseudo-terminal restriction fragments, a PCR-related bias affecting terminal restriction fragment length polymorphism analysis of microbial community structure. *Applied and Environmental Microbiology* **69**, 2555-2562.
- Fierer N, Jackson RB (2006) The diversity and biogeography of soil bacterial communities. *Proceedings of National Academy of Science USA* **103**, 626-631
- Goodfellow M, Williams ST (1983) Ecology of actinomycetes. *Annual Reviews in Microbiology* **37**, 189-216.
- Griffiths RI, Whiteley AS, O'Donnell AG, Bailey MJ (2000) Rapid method for coextraction of DNA and RNA from natural environments for analysis of ribosomal DNA- and RNA-based microbial community composition. *Applied and Environmental Microbiology* **66**, 5466-5491.
- Mader P, Fliebbach A, Dubois D, Gunst L, Fried P, Niggli U (2002) Soil fertility and biodiversity in organic farming. *Science* **296**, 1694-1697.
- McCarthy AJ, Williams ST (1990) Methods for studying the ecology of actinomycetes. *Methods in Microbiology* **22**, 533-563
- O'Donnell AG, Seasman M, Macrae A, Waite I, Davies JT (2001) Plants and fertilisers as drivers of change in microbial community structure and function in soils. *Plant and Soil* **232**, 135-145.

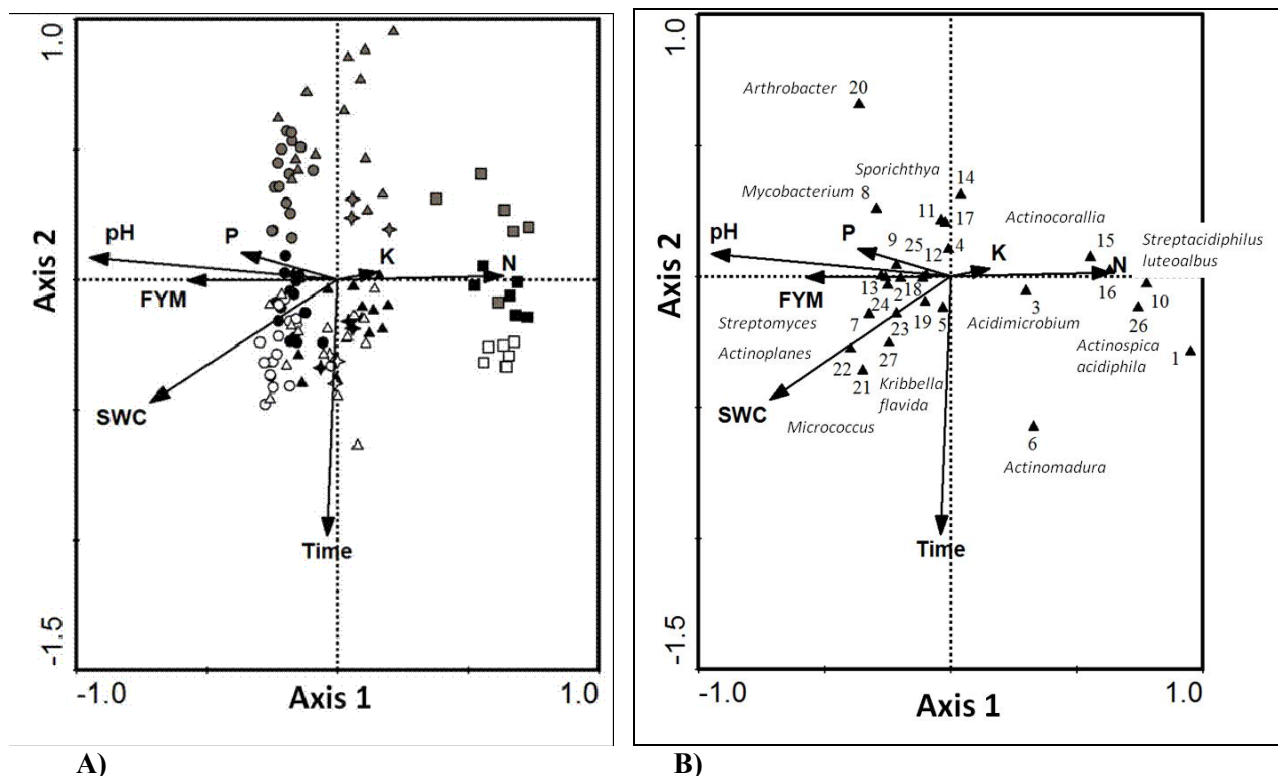


Figure 1. Bi-plots from canonical correspondence analysis (CCA) showing the relationship between a) actinobacterial community composition (as judged by T-RFLP profiles), fertiliser management and time of sampling at Palace Leas [first two axes account for 44% of total variance, axis 1 (32%) and axis 1 (12%)]. Points on the graph represent actinobacterial communities sampled from the soil plots under different management regimes [plots 1-5 receive organic inputs (●) plots 8, 9, 10, 12 & 13 receive mineral fertilisers inputs including P (▲), plots 7 & 11 receive mineral fertiliser inputs without P (■) and plot 6 the control receives no fertiliser inputs (◆)]. Arrows represent the environmental variable [inorganic inputs (FYM), inorganic inputs (N, P, K), pH, soil water content (SWC) and time of sampling (Time)]. In order to assess temporal changes in community structure throughout the year the May (grey ■), September (black ■) and January (white □) samples were assigned as 0, 4 and 8 months, respectively, which represents the number of months after initial sampling in May.

b) the distribution of different taxa (as judged by terminal restriction fragments, TRFs) with management and time of sampling. Triangles on the graph represent TRFs derived from T-RFLP analysis on the soil plots under different management regimes at Palace Leas.

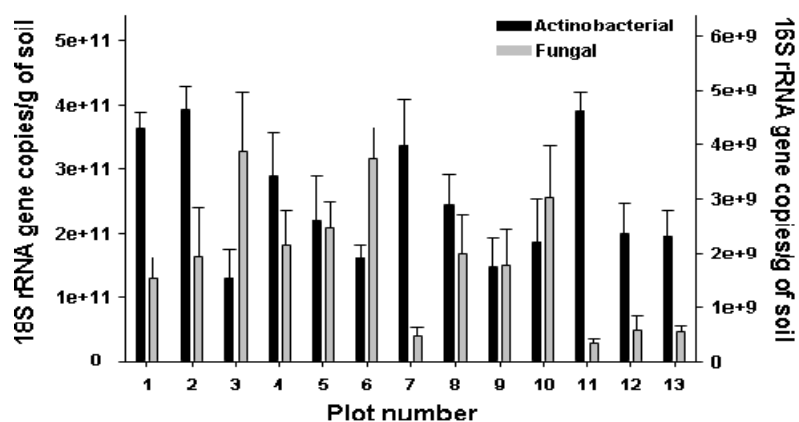


Figure 2. Actinobacterial and fungal abundances in the 13 plots at the Palace Leas Hay Meadow Experiment estimated using real time qPCR of 16S rRNA and 18S rRNA genes respectively. Plots 1-5 have received organic inputs (FYM) either singly or in combination with inorganic fertilisers since 1896. Plots 7 to 13 have received only inorganic fertilisers over the same period. The control (plot 6) has received no fertiliser input since 1896. All qPCR reactions were done in triplicate on separate soil samples. Error bars indicate the standard error where n=3.

Alteration of soil chemical properties by two species of millipedes

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Abstract

Two species of millipedes, i.e. *Parafontaria laminata* and *Parafontaria tonominea* swarm every 8 years and every 3 years, respectively. Both of them are in the group of large species with 3-5cm length among the millipedes. Influence of the millipedes through the activities such as feeding and discharge on soil physicochemical properties and decomposition of organic matter are relatively large among the soil animals because of their emergence of high density and large body size. We investigated differences of chemical properties of the casts of these two species millipedes to clarify how they alter the soil properties and how they differ between two species. The millipedes were fed for 1 week at four treatments with contrasting soil type, leaf litter, and density of millipede in a laboratory experiment. After one week feeding, we collected their casts and analyzed (TC, TN, ignition loss, and soil respiration rate). Both millipedes ingested leaf litter and soil. The results were that (1) *P. laminata* fed on more soil than *P. tonominea* did. (2) Higher levels of geophagy were observed in high-density *P. tonominea* treatments. (3) As rate of feeding of leaf litter increased, soil respiration increased.

Key Words

Soil animal, ecosystem engineer, Diplopoda, cast, decomposition rate, chemical properties.

Introduction

Activities of macrofauna such as millipede and earthworm affect the soil ecosystem. They are called litter transformers and ecosystem engineers (Kaneko *et al.* 2004). Swarmings of two species of *Parafontaria laminata* (Attems 1909) and *Parafontaria tonominea* Attems were observed around Mt. Yatsugatake and Mt. Sanbe in 2008, respectively. *Parafontaria* spp. are large millipedes. The body size of adult *P. laminata* is about 3-4cm length and about 0.3g in wet weight, and adult *P. tonominea* is 4-5cm length and about 0.5g in wet weight. Periodic swarming by adult *P. laminata* and *P. tonominea* has occurred on an 8-year cycle (Nijima *et al.* 1988) at Central Japan and a 3-year cycle at Mt. Sanbe in Shimane prefecture, respectively. Their activities such as mass transfer by their moving and their synchronous feeding on soil and litter might have considerably affected soil ecosystem because of emergence of these adults in very high population densities. Nijima (1984) reported that the surface soil in the habitat area of *P. laminata* contained high amounts of organic matter than in soil without the millipede because the cast is high in organic matter. *P. laminata* increased nitrogen mineralization with discharge excretion as soil aggregates (Okai *et al.* 2008). Fujimaki *et al.* (2009) found that it is difficult to decompose organic matter of inside in their casts. Furthermore, Kaneko (1999) reported that ingestion of litter and soil by *P. tonominea* also increased soil respiration and leaching of Ca^{2+} , Mg^{2+} and nitrate.

"What do millipedes ingest?" and "What type of casts the millipedes discharge?" are important information in respect to change of soil quality and a material cycle in soil ecosystem. In order to clarify these matters in the present study, we focused on foods and casts of two species of millipedes, and investigated the chemical properties of their casts which we got by feeding them in the laboratory experiment under several feeding conditions.

Materials and Methods

Adults of *P. laminata* and *P. tonominea* were collected at around Mt. Yatsugatake (35°58'N, 138°22'E), Yamanashi prefecture, central Japan on beginning of October 2008, and Mt. Sanbe (35°08'N, 132°37'E), Shimane prefecture, western Japan on the end of September 2008, respectively.

The millipedes were fed for 1 week in the dark in 20°C at four treatments (Table 1): (A) train millipede with Andisol and larch leaf litter in Yatsugatake, (B) train millipede with Andisol and leaf litter of deciduous broad-leaved dominated by Oak in Mt. Sanbe, (C) *P. tonominea* with Andisol and leaf litter of deciduous broad-leaved dominated by Oak in Mt. Sanbe, and (D) high density of *P. tonominea* with Andisol and leaf litter of deciduous broad-leaved dominated by Oak in Mt. Sanbe. Soil was placed 3 cm deep in the bottom of a plastic box (31.6 x 43cm), and litter was put on the soil surface (Figure 1). Soil was sieved with 2 mm mesh. Soil water content was adjusted about 60% of the maximum water holding capacity of the soil. Litter water content was also adjusted about 50%. Millipedes were added in the plastic boxes at (A) (B) 588-662 individuals/m², (C) 324 individuals/m², and (D) 810 individuals/m². After 1 week, casts of millipedes were collected from plastic boxes, and the litter removed. Samples of more than 2 mm diameter were used as casts. We analyzed the soil and the cast for total carbon (TC), total nitrogen (TN), ignition loss, and soil respiration by incubation for 8weeks at 25°C.

Table 1. Four feeding treatments of the combination of soil, leaf litter and density of millipede.

Table 1. Four recycling treatments of the combination of soil, leaf litter and density of millipede.							
Soil (Andisol)			Leaf Litter		Millipede		
	Area	Dry weight (kg / box)	Forest type and Area	Dry weight (g / box)	Species	Number (Density m ⁻²)	Biomass (g /box)
A	Yatsugatake	1.2	Larch Yatsugatake	76	<i>Parafontaria laminata</i>	82-90 (603-662)	26.3-28.9
B	Sanbe	1.3	Broadleaves Sanbe	52	<i>Parafontaria laminata</i>	82 (603)	26.5
C	Sanbe	1.3	Broadleaves Sanbe	52	<i>Parafontaria tonominea</i>	44 (324)	23.0
D	Sanbe	1.3	Broadleaves Sanbe	52	<i>Parafontaria tonominea</i>	104-110 (765-810)	51.9-54.7

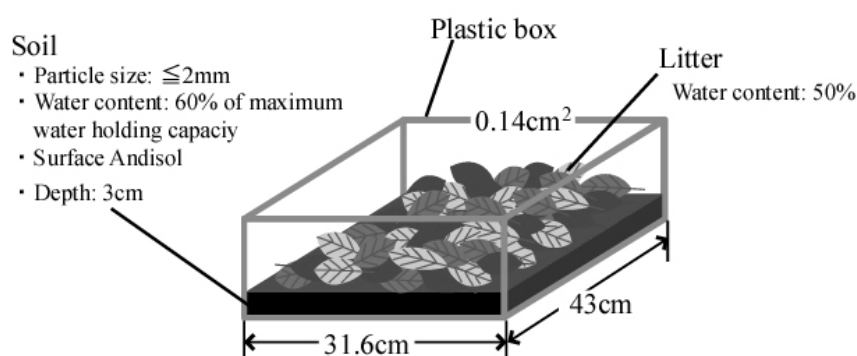


Figure 1. Experimental design of a feeding box and soil and litter condition.

Results and Discussion

Values of all measuring items (TC, TN, ignition loss and soil respiration) of cast of millipedes were higher than those of original soils in Mt. Yatsugatake and Mt. Sanbe (Table 2 and Figure 2).

Table 2. TC, TN and ignition loss of cast of millipedes and original soils and litters

Material area		Total C (%)	Total N (%)	C/N	Ignition loss (%)
Mt. Yatsugatake	Soil	20.63	1.20	17.23	44.55
	Casts of <i>P. laminata</i> (A)	24.58	1.29	19.03	51.58
	Larch litter	48.56	1.56	31.22	
Mt. Sanbe	Soil	13.82	0.81	16.96	30.45
	Casts of <i>P. laminata</i> (B)	18.91	1.00	18.92	40.58
	Casts of <i>P. tonominea</i> (C)	21.40	1.06	20.25	46.21
	Casts of <i>P. tonominea</i> (D)	19.73	1.04	19.03	42.66
	Broadleaves litter	50.21	1.62	31.03	

Table 2 shows TC and TN values in soils, these were higher in Mt. Yatsugatake than in Mt. Sanbe. These values in litter in both sites were little different. The casts of both species of millipedes in all treatments

showed TC and TN values higher than that of the soils and less than that of the litters. These results indicated that both species of millipedes fed on litter and soil. TC and TN values of the cast increased in order of treatment (B), (D) and (C). Although there were no quantitative data, it was observed that *P. laminata* (B) preferably fed on soil to litter compared with *P. tonomineae* (C) and (D) did. From these facts and observation, we assumed that TC and TN increased as the rate of ingestion of leaf litter increased. Ignition loss also increased in order of treatment (B), (D) and (C) as the feeding proportion of litter against to soil increased (Table 2). Ignition loss indicates organic matter content in the casts. Organic matter content of the casts depended on their feeding proportion of soil and litter.

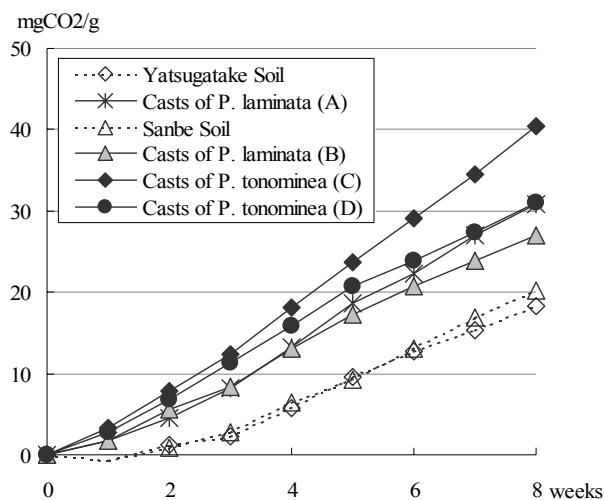


Figure 2. Soil respiration from soil and casts of millipedes by incubation for 8 weeks at 25°C

The results of TC, TN and ignition loss of *P. tonomineae* (C), which is same biomass with *P. laminata* (B) and lower density than *P. tonomineae* (D), were higher than (B) and (D) (Table 2). These results meant that *P. laminata* fed on more soil than *P. tonomineae* and higher levels of geophagy were observed in high-density *P. tonomineae* treatments. Hashimoto *et al.* (2004) reported that *P. laminata* increased their geophagy rate as the population density increased.

Comparing the data between (B), (C) and (D) that the millipedes fed under soil and leaf litter from Mt. Sanbe on soil respiration, we found that as the millipedes fed more on litter increased, soil respiration increased (Figure 2). In general, low C/N ratio of organic matter is easily decomposed. Though C/N ratio of the casts (C) is higher than the casts (B), soil respiration from the casts (C) is higher than (B). This result indicated that the casts (C) probably contained much organic matter derived from leaf litter. Species of millipedes differed their feeding rate of soil and leaf litter. The differences of their living density seemed to effect on decomposition rate of organic matter.

References

- Fujimaki R, Tayasu I, Asanuma H, Okai N, Sato Y, Kaneko N (2009) Relations of habitat history of litter-soil feeder and accumulation of organic matter in forest soil: consideration from effect on soil alteration. Proceedings of the 56th annual meeting of ecological society of Japan, p.442 (in Japanese).
- Hashimoto M, Kaneko N, Ito MT, Toyota A (2004) Exploitation of litter and soil by the train millipede *Parafontaria laminata* (Diplopoda: Xystomidae) in larch plantation in Japan. *Pedbiologia* **48**, 71-81.
- Kaneko N (1999) Effect of millipede *Parafontaria tonomineae* Attems (Diplopoda: Xystomidae) adult on soil biological activities: microcosm experiment. *Ecological Research* **14**, 271-279.
- Kaneko N, Ito MT (2004) Biodiversity and ecosystem function of soil animals. *Japanese Journal of Ecology* **54**, 201-207 (in Japanese).
- Niijima K (1984) The outbreak of the train millipede. *Japanese Journal of Forest Environment* **26**, 25-32 (in Japanese).
- Niijima K, Shinohara K (1988) Outbreaks of the *Parafontaria laminata* group (Diplopoda: Xystomidae). *Japanese Journal of Ecology* **38**, 257-268 (in Japanese with English summary).
- Okai N, Fujimaki R, Sato Y, Kaneko N (2008) Influence of feeding of larvae train millipede *Parafontaria laminata* on acceleration of nitrogen mineralization. Proceedings of the 31th annual meeting of the Japanese society of soil zoology, p.16 (in Japanese).

Anecic earthworms and associated ecosystem services in a ley-arable crop rotation

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Abstract

Earthworms in general and anecic earthworms in particular, play a key role in the ecosystem service of water regulation through enhancing water infiltration and stimulating root growth to deeper soil layers by earthworm burrows. A crop rotation of grass and maize can be sustainable in terms of efficient nutrient use, but there is lack of information on the effect on earthworms. Earthworms were sampled over three years in a 36 years old experiment. Permanent arable land was compared with permanent grassland and with a ley-arable crop rotation. In the first year of arable cropping in the rotation, the number of earthworms was already low and not different from continuous cropping. In the three-year grass ley, the abundance of earthworms returned to the level of permanent grassland in the second year. The restoration of earthworm biomass took a minimum of three years. However, the anecic species did not recover in the three-years grass ley to the dominance they had in the permanent grassland. The number of earthworm burrows was related to earthworm biomass and was highest in permanent grassland. Our data suggest that anecic earthworms are under pressure in a ley-arable crop rotation, which may have a negative impact on the ecosystem service of water regulation.

Key Words

Ecosystem services, earthworms, crop rotation, grassland, water regulation.

Introduction

The rainfall in the Netherlands has increased in the last 100 years by 18% and the number of days with more than 50 mm of rainfall has increased in the last 50 years from 5.4 to 9.0 (Anonymous, RNMI; Boxel and Cammeraat, 1999). On the other hand the length of drought periods in summer increases. Therefore, the ecosystem service of water regulation by soil biota becomes more important. Earthworms play a key role in water regulation (Hoogerkamp et al., 1983; Clements et al., 1991): epigeic and endogeic earthworms increase water infiltration in the topsoil through their burrowing activity, while the deep, vertical burrows of anecic earthworms increase water infiltration and root growth (Logsdon and Linden, 1992; Edwards and Shipitalo, 1998). Bouché and Al-Addan (1997) measured an average infiltration rate of 282 mm h⁻¹ per 100 g m⁻² for anecic species and 150 mm h⁻¹ per 100 g m⁻² of all the other earthworms.

Recent legislative restrictions on the use of organic and artificial N fertilizers and a quest for sustainable farming systems, have brought attention back to crop rotations with grass and maize. A crop rotation of grass and maize can be sustainable in terms of efficient nutrient use (Nevens and Reheul 2002; 2003). Furthermore, ley farming guarantees a high clover content and provides an opportunity to control perennial weeds on organic farms (Younie and Hermansen 2000). However, there is lack of information on the effect of such a crop rotation on abundance and ecological group composition of earthworms.

Earthworms were sampled over three years in a 36 years old experiment in which permanent arable land was compared with permanent grassland and with a ley-arable crop rotation. Our objectives were (1) to determine the long-term effects of a ley-arable crop rotation on earthworms in comparison with permanent grassland and continuous arable cropping, and (2) to assess the short-term recovery of soil biota in a ley-arable crop rotation.

Methods

Sampling site and experimental design

In 1966, a crop rotation experiment was established on a sandy loam soil at the experimental farm of Ghent University at Melle (50° 59'N, 03°49'E; 11 m above sea level) (Nevens and Reheul 2001; 2003). Four

treatments were established in a complete randomised block design with four blocks:

PG: Permanent grassland since 1966;

TG: Temporary ley-arable crop rotation, started in 1966 with three years of grass ley followed by three years of arable land cropped with forage crops;

TA: Temporary arable crop-ley rotation. This treatment is comparable to TG but started in 1966 with three years of arable cropping followed by three years of grass ley;

PA: Permanent arable cropping since 1966.

In the seventh rotation of the trial the TG treatment was established in April 2002 after rotavating the maize stubble of the preceding three years of arable cropping. The seed mixture used was 40 kg *Lolium perenne* L. ha⁻¹ (cvs. Plenty and Roy) and 4 kg *Trifolium repens* L. ha⁻¹ (cv. Huia). In the seventh rotation the TA treatment was established in 2002 after rotavating the former grass ley on 9 April. In addition to nitrogen fertilizer (ammonium nitrate 27%), all plots received a basal fertilizer application of triple super phosphate (45% P₂O₅) and potassium chloride (40% K₂O). Herbicides were used in the treatments with maize, according to good agricultural practices. No pesticides were used on the grassland plots.

Earthworm sampling

In the first three years of the seventh rotation, earthworms were sampled in 2 blocks (20x20x20 cm) per plot on 30 October 2002, 7 October 2003 and 15 October 2004. The blocks were transferred to the laboratory where the earthworms were hand-sorted, counted, weighed and fixed in alcohol prior to identification. Numbers and biomass were expressed per m². Adults were identified according to species. A distinction was made between (1) epigeic species (pigmented, living superficially in the litter layer, little burrowing activity), (2) endogeic species (living in burrows at approximately 10-15 cm depth) and (3) anecic species (relatively large worms, living in vertical burrows from which they collect dead organic matter at the surface at night) (Bouché 1977). In 2004, before the blocks were sorted for earthworms, the earthworm burrows with a diameter >2 mm were counted on horizontal surfaces (20x20 cm) at 10 cm and 20 cm depth.

Statistical analysis

The data were analysed with GENSTAT (8th Edition, VSN International, Hemel Hempstead, UK) using a two-way ANOVA in randomised blocks with treatment (PG, TG, TA and PA) and year of sampling as factors.

Results

The number of earthworms was highest in the PG treatment followed by the TG treatment (Table 1). On arable land (TA and PA) the number of earthworms was 12-24% of PG. The significant interaction of treatment and year is mainly due to the recovery in the number of earthworms in the TG treatment (Figure 1). In October 2003, the second year after the establishment of grass (TG) in the arable-ley crop rotation, the number of earthworms reached the same level as in the PG treatment.

The body biomass of the earthworms in TG was significantly lower ($P < 0.001$) than in PG, and therefore the recovery of the total biomass was not as spectacular as the total numbers. Even in October 2004, the final year of the three-year period of grass ley, the earthworm biomass in TG was different from PG: 96 g m⁻² compared to 163 g m⁻² ($P = 0.002$). In total numbers and biomass the TA plots resembled the PA plots. Numbers and especially biomass in the TA treatment already reached a low level in the first year of the rotation, suggesting a rapid decrease in earthworms after rotavating the grass ley.

Species of earthworms found in the trial were *Lumbricus rubellus*, *Aporrectodea caliginosa*, *Allolophora chlorotica*, *Aporrectodea rosea* and *Aporrectodea longa*. PG had the highest number of species and the arable treatments the lowest number. The interaction between treatment and year was mainly due to an increase of the number of species in the TG treatment from 2002 to 2003. As with the total numbers, the number of species in TG almost recovered within two years in comparison with the PG treatment. Among the adult earthworms in the PG plots, the anecic species were dominant (52 % anecic species). In TG and the arable treatments (TA and PA), the endogeic species were most common: 62 %, 88 % and 100 %, respectively. The epigeic species were mainly found in the grass treatments.

The number of earthworm burrows showed a clear decrease in the order PG>TG>TA>PA (Table 1). The variation in the number of burrows at 10 cm depth, measured in October 2004, was explained by a regression

model ($R^2=0.93$) with treatment and earthworm biomass measured in 2004 as fitted terms (Figure 2). There was a significant positive relation between the biomass and the number of burrows for all treatments (slope=0.058, $P=0.044$). In the model all four individual treatment levels differed significantly ($P<0.001$).

Table 1. Earthworm numbers, biomass, species, functional groups and earthworm burrows in permanent grassland (PG), temporary grassland (TG), temporary arable land (TA) and permanent arable land (PA): averages from three consecutive years (2002-2004).

Earthworms	Units	Treatments				P-value	Year P-value	Treat.*year P-value
		PG	TG	TA	PA			
Total number	$n\ m^{-2}$	256 a	187 b	62 c	30 c	<0.001	NS	0.008
Body biomass	$g\ worm^{-1}$	0.65 a	0.25 b	0.23 b	0.12 b	<0.001	0.033	NS
Total biomass	$g\ m^{-2}$	166 a	52 b	14 bc	5 c	<0.001	NS	NS
Number of species	$n\ 20\times20\times20cm^{-1}$	2.0 a	1.3 b	0.5 c	0.2 c	<0.001	NS	<0.001
Epigeic adults	$n\ m^{-2}$	20 a	25 a	1 b	0 b	0.016	0.011	NS
Endogeic adults	$n\ m^{-2}$	46 ab	49 a	22 bc	7 cd	0.009	NS	0.031
Anecic adults	$n\ m^{-2}$	71 a	4 b	2 b	0 c	<0.001	NS	NS
Earthworm burrows*								
10 cm depth	$n\ m^{-2}$	388 a	238 b	106 c	6 d	<0.001	--	--
20 cm depth	$n\ m^{-2}$	356 a	206 b	100 c	6 d	<0.001	--	--

* Earthworm burrows were counted in 2004 only

Values followed by the same letter within a row are not statistically different at the 5% error level for the main treatment effect.

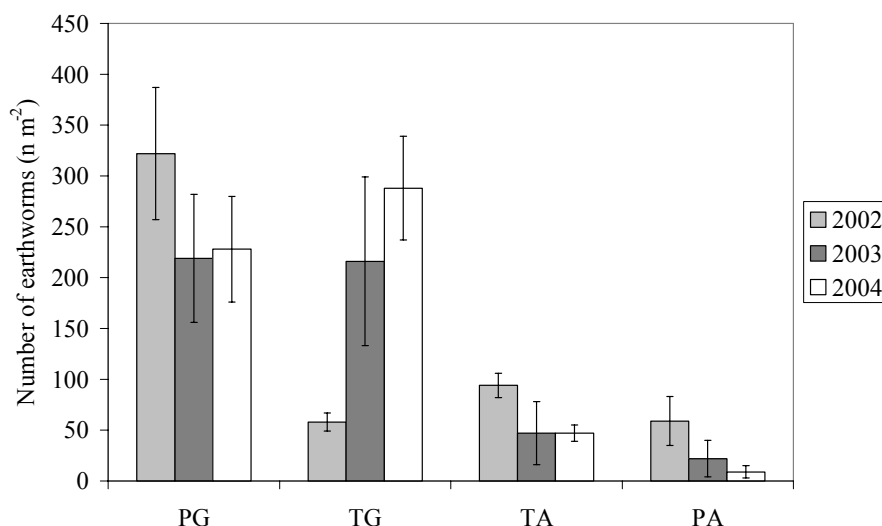


Figure 1. Mean number (\pm SE) of earthworms ($n\ m^{-2}$) in permanent grassland (PG), temporary grassland (TG), temporary arable land (TA) and permanent arable land (PA) in the 7th rotation of a 36 years old experiment

Discussion and conclusions

It is well known that grassland contains more earthworms than arable land (Edwards and Bohlen 1996). In this experiment, the number of earthworms in the PA treatment was as low as 12% of the number in the PG treatment. Edwards and Bohlen (1996) mention two reasons for a decreased number of earthworms, besides the mechanical damage and predation after cultivation: the loss of an insulating layer of vegetation and a decreased food supply. The small number of earthworms in the TA treatment, six months after rotavating the grass ley, suggests that the decrease in earthworm numbers in our experiment was rapid. Growing grass over several years favours the growth of earthworm populations and the best way of maintaining a large earthworm fauna in agricultural land is to include ley farming (Edwards and Bohlen 1996). In fact, in the ley phase of this experiment, earthworm biomass increased from $8\ g\ m^{-2}$ in the first year to $51\ g\ m^{-2}$ in the second year and to $96\ g\ m^{-2}$ in the third year. This is a biomass increase of $40\text{--}45\ g\ m^{-2}$ per year. Assuming a temporal constant increase, the grass ley were to last for 4 to 5 years in order to reach similar biomass levels as found in PG. A more lasting difference between the PG treatment and the remaining ley-arable crop rotation treatments, is the dominance of the anecic species in PG. The data suggest that especially anecic earthworms are under pressure in a ley-arable crop rotation which may have a negative impact on the ecosystem service of water regulation under future grassland.

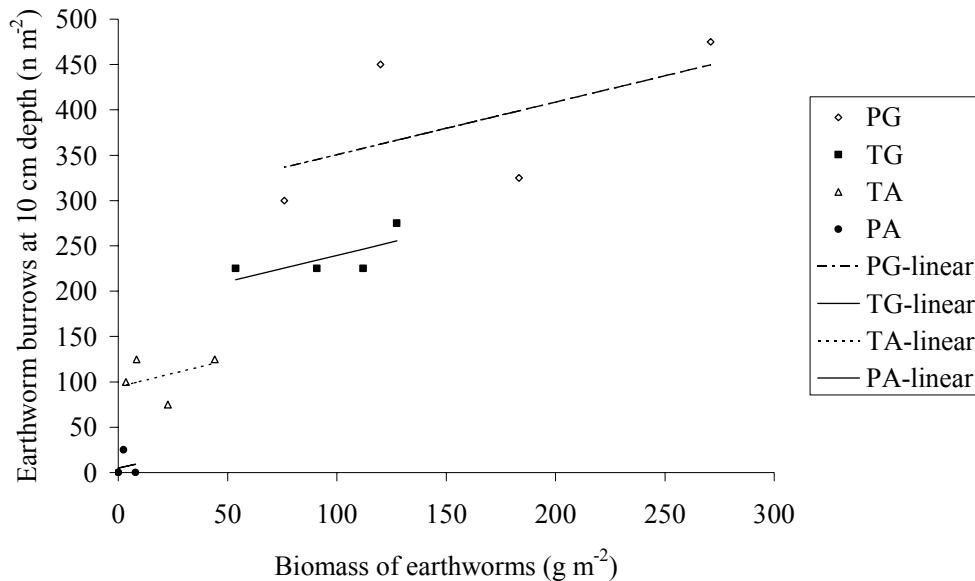


Figure 2. Relation between earthworm biomass (g m^{-2}) and earthworm burrows at 10 cm depth (n m^{-2}) in 2004 for the four treatments ($R^2=0.93$). Earthworm burrows (n m^{-2} at 10 cm depth) = treatment ($P<0.001$) (intercept 5 for PA, 95 for TA, 182 for TG, 293 for PG) + $0.58 \times$ earthworm biomass (g m^{-2}) ($P=0.044$). PG=permanent grassland, TG=temporary grassland, TA=temporary arable land, PA=permanent arable land.

References

- Anonymous, Royal Netherlands Meteorological Institute, Risk analysis of heavy rain.
- Bouché MB (1977) Strategies lombriciennes. In 'Soil organisms as Components of Ecosystems'. (Eds U Lohm, T Persson) pp122-132. *Ecological Bulletins* **25**
- Bouché MB, Al-Addan F (1997) Earthworms, water infiltration and soil stability: some new assessments. *Soil Biol. Biochem.* **29**, 441-452.
- Boxel JH, Cammeraat J (1999) Een analyse van de neerslag in deze eeuw; Wordt Nederland steeds natter? *Meteorologica* **8** (1), 11-15.
- Clements RO, Murray PJ, Sturdy RG (1991) The impact of 20 years' absence of earthworms and three levels of N fertilizers on a grassland environment. *Agric. Ecosyst. Environ.* **36**, 75-85.
- Edwards CA, Bohlen PJ (1996) *Biology and Ecology of Earthworms*, 3rd edn., Chapman and Hall, London, 426 pp.
- Edwards WM, Shipitalo MJ (1998) Consequences of earthworms in agricultural soils: aggregation and porosity. In 'Earthworm Ecology'. (Eds CA Edwards) pp. 147-161 (St Lucie Press, Boca Raton, FL)
- Hoogerkamp M, Rogaar H, Eijssackers HJP (1983) Effects of earthworms on grassland on recently reclaimed polder soils in the Netherlands. In: Satchell JE (Eds.), *Earthworm Ecology: from Darwin to vermiculture*, Chapman and Hall, London, 85-105.
- Logsdon SD, Linden RD (1992) Interactions of earthworms with soil physical conditions influencing plant growth. *Soil Science*, **154**(4), 330-337.
- Nevens F, Reheul D (2001) Crop rotation versus monoculture; yield, N yield and ear fraction of silage maize at different levels of mineral N fertilization. *Neth. J. Agric. Sci.* **49**, 405-425.
- Nevens F, Reheul D (2002) The nitrogen- and non-nitrogen-contribution effect of ploughed grass leys on the following arable forage crops: determination and optimum use. *Eur. J. Agron.* **16**, 57-74.
- Nevens F, Reheul D (2003) Permanent grassland and 3-years leys alternating with 3 years of arable land: 31 years of comparison. *Eur. J. Agron.* **19**, 77-90.
- Van Eekeren N, Bommelé L, Bloem J, Rutgers M, De Goede R, Reheul D, Brussaard L (2008) Soil biological quality after 36 years of ley-arable cropping, permanent grassland and permanent arable cropping. *Appl. Soil Ecol.* **40**, 432-446.
- Younie D, Hermansen J (2000) The role of grassland in organic livestock farming. In 'Grassland Farming-Balancing environmental and economic demands, Proceedings of the 18th General Meeting of the European Grassland Federation (Eds. Søgaard et al.) pp. 493-509 (Aalborg, Denmark)

Carbon input and soil carbon dioxide emission affected by land use and management practices

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Abstract

Land use and management practices may influence carbon inputs and soil CO₂ emissions, a greenhouse gas responsible for global warming. C inputs and soil CO₂ emissions were monitored from crop and grassland with various irrigation and cropping systems from 2006 to 2008 in western North Dakota, USA. Treatments were two irrigation systems (irrigated and non-irrigated) and six cropping systems [conventional-tilled barley with N (CTBFN), conventional-tilled barley with no N (CTBON), no-tilled barley-pea with N (NTB-PN), no-tilled barley with N (NTBFN), no-tilled barley with no N (NTBON), and no-tilled Conservation Reserve Program (grassland) (NTCRP)]. Crop residue C was greater in irrigated than in non-irrigated systems and greater in CTBFN and NTBFN than in other cropping systems. Soil CO₂ flux varied with time of measurement in various irrigation and cropping systems. Total CO₂ flux from May to October was greater in irrigated than in non-irrigated systems and greater in NTCRP than in other treatments. Differences in crop C inputs, root and soil respiration, and soil temperature and water content can result in variations in CO₂ emissions among management practices and land use.

Key Words

Carbon dioxide, cropping system, irrigation, nitrogen fertilization, residue carbon, tillage.

Introduction

Emission of CO₂ from soils under cropland and grassland contributes a significant source of greenhouse gas responsible for global warming (Duxbury 1994). The emission occurs due to root and soil respiration and organic matter mineralization (Curtin *et al.* 2000; Sainju *et al.* 2008). In contrast, soil is also an important sink of atmospheric CO₂ which is absorbed by plant biomass through photosynthesis and converted into soil organic matter after the residue is returned to the soil (Lal *et al.* 1995). Management practices, such as irrigation, tillage, cropping system and N fertilization can alter crop residue C inputs, nutrient dynamics, and soil temperature and water contents that influence soil surface CO₂ emissions (Curtin *et al.* 2000; Al-Kaisi and Yin 2008; Sainju *et al.* 2008).

Materials and methods

The experiment was conducted in western North Dakota in a sandy loam soil. Treatments consisted of two irrigation systems (irrigated and non-irrigated) as main plot and six cropping systems (CTBFN, CTBON, NTB-PN, NTBFN, NTBON, and NTCRP) as split-plot treatments in a randomized block design with three replications. Malt barley and pea were planted in April and harvested in October. The NTCRP treatment consisted of mixed alfalfa and grasses (crested wheatgrass and western wheatgrass) that were self regenerated. Plant biomass was measured from two 0.5 m² areas and crop grain yield was determined with a combine harvester. After harvesting grains in croplands, biomass residue (stems + leaves) were returned to the soil. In NTCRP treatment, grass biomass was allowed to self recycle in the soil. Soil surface CO₂ flux was measured from 9 A.M. to 12 A.M. every week from April to October, 2006 to 2008, with an Environmental Gas Monitor chamber containing an infrared CO₂ analyzer attached to a data logger (PP System, Haverhill, Massachusetts, USA). The chamber was placed at the soil surface for 2 min in each plot until CO₂ flux measurement was recorded in the data logger. At the time of measurement, soil temperature at the 0-15 cm depth was measured with a probe attached to the data logger and soil water content was determined gravimetrically by collecting field-moist soil sample and oven drying at 105°C.

Results and discussion

Crop residue C differed significantly among treatments (Table 1). Under irrigated condition, residue C was greater in CTBFN than in other cropping systems, except in NTBFN. Under non-irrigated conditions, residue C was greater in CTBFN, CTBON, and NTBFN than in NTBON and NTCRP. Averaged across cropping systems, residue C was greater in irrigated than in non-irrigated systems. Similarly, averaged across

Table 1. Effect of irrigation and cropping system on crop residue C input averaged across years.

Cropping system†	Irrigation (Mg C/ha)		
	Irrigated	Non-irrigated	Mean
CTBFN	2.42aA‡	1.72aB	2.07a
CTBON	1.31cA	1.40aA	1.36c
NTB-PN	1.88bA	1.35abB	1.61b
NTBFN	2.25abA	1.63aB	1.94a
NTBON	1.01cA	0.98bcA	0.99d
NTCRP	1.12cA	0.69cB	0.90d
Mean	1.67A	1.29B	
<u>Contrasts</u>			
Till vs. no-till			0.25*
N fertilization vs. no N fertilization.			0.83***
Cont. barley vs. barley-pea in no-till			0.33*
Cropping vs. grasses in no-till			0.69***

irrigation system, residue C was greater in CTBFN and NTBFN than in other cropping systems. Residue C was greater with tillage than with no-tillage, greater with N fertilization than without, greater with continuous barley than barley-pea, and greater with crops than with grasses. Soil surface CO₂ flux varied with measurement dates from April to October 2007 among cropping systems (Figure 1). The flux was normally greater with NTCRP than with other cropping systems from April to July. Greater fluxes in all cropping systems were observed immediately following substantial rainfall in June and July (Figure 2), a result of increased microbiological activity due to increased soil water content and temperature (Figure 1). Similar results were obtained in 2006 and 2008. The flux was also greater with irrigation than without at most measurement dates, probably due to increased water availability.

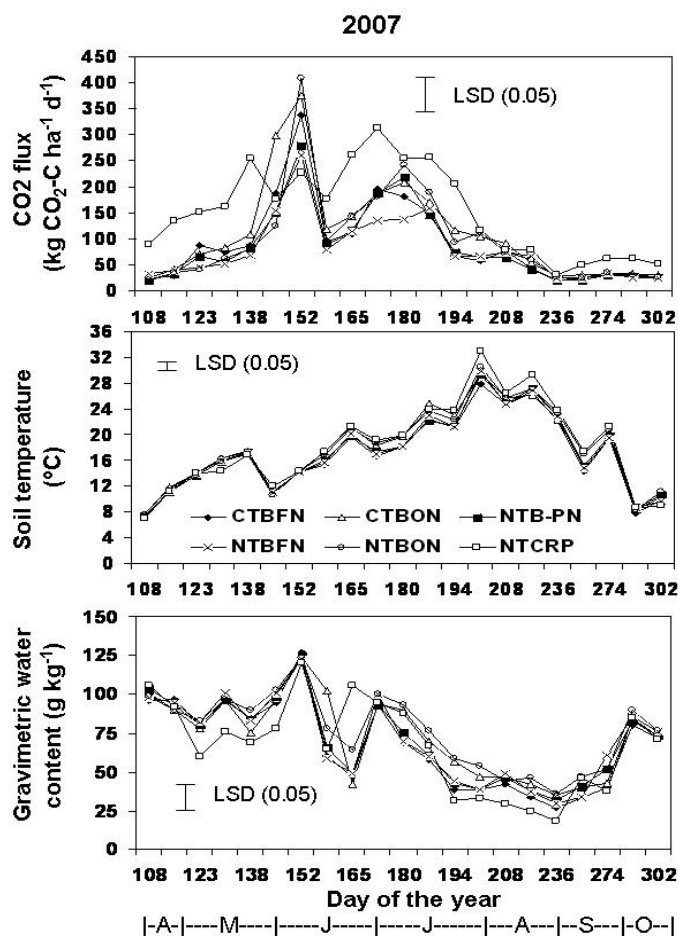


Figure 1. Effect of cropping system on soil surface CO₂ flux and soil temperature and water content at the 0-15 cm depth in 2007. CTBFN, conventional-tilled malt barley with 67 to 134 kg N/ha; CTBON, conventional tilled malt barley with 0 kg N/ha; NTB-PN, no-tilled malt barley-pea rotation with 67 to 134 kg N/ha applied to malt barley; NTBFN, no-tilled malt barley with 67 to 134 kg N/ha; NTBON, no-tilled malt barley with 0 kg N/ha; and NTCRP, no-tilled Conservation Reserve Program.

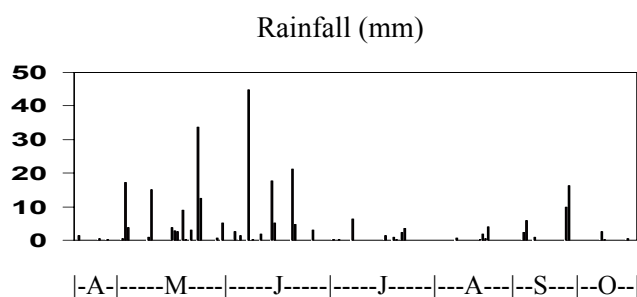


Figure 2. Rainfall distribution during measurement dates in 2007 (total rainfall from May to October 2007 = 278 mm).

Total CO₂ flux from April to October was greater in NTCRP than in other cropping systems in all years (Table 2). The flux was lower in NTBFN and NTB-PN in 2 out of 3 years. Tillage increased the flux compared with no-tillage in 2 out of 3 years but N fertilization decreased the flux compared with no N fertilization in 1 out of 3 years. The greater CO₂ flux but lower or similar soil water content in NTCRP than in other cropping systems (Table 2, Figure 1) suggests that grass roots in their undisturbed condition continued to respire from April to July when crops were still growing. Although C inputs from aboveground residue was lower in NTCRP than in other cropping systems (Table 1), belowground (root) residue was considered to be greater with grasses than with crops. As a result, increased root respiration and mineralization of belowground residue could have increased CO₂ flux in NTCRP. In croplands, assuming root growth was similar to aboveground growth; increases in crop residue C did not always increase CO₂ flux. Rather flux was greater with CTBON than with other treatments, probably due to tillage (Curtin *et al.* 2000). Nitrogen fertilization has been known to decrease CO₂ flux (Al-Kaisi and Yin 2008).

Table 2. Effect of irrigation and cropping system on soil surface CO₂ flux at the 0- to 15-cm depth from 2006 to 2008.

Irrigation	Cropping system†	Soil surface CO ₂ flux (kg CO ₂ -C/ha/d)		
		2006	2007	2008
Irrigated		132a‡	118a	148a
Non-irrigated		88b	91b	111b
	CTBFN	96c	93cd	131bc
	CTBON	112b	112b	147b
	NTB-PN	96c	87d	116cd
	NTBFN	96c	82d	118cd
	NTBON	100bc	101bc	110d
	NTCRP	161a	153a	161a
Contrasts				
	Till vs. no-till	12	22*	49***
	N fertilization vs. no N fertilization.	-19	-38**	-8
	Cont. barley vs. barley-pea in no-till	2	5	-2
	Cropping vs. grasses in no-till	-64***	-63***	-46**

† Cropping systems are CTBFN, conventional-tilled malt barley with 67 to 134 kg N/ha; CTBON, conventional tilled malt barley with 0 kg N/ha; NTB-PN, no-tilled malt barley-pea rotation with 67 to 134 kg N/ha applied to malt barley; NTBFN, no-tilled malt barley with 67 to 134 kg N/ha; NTBON, no-tilled malt barley with 0 kg N/ha; and NTCRP, no-tilled Conservation Reserve Program.

‡ Numbers followed by different lower case letters within a column and upper case letter within a row in a set are significantly different at $P \leq 0.05$ by the least significant difference test.

*, **, and *** Significant at $P \leq 0.05$, 0.01, and 0.001, respectively.

Conclusions

Aboveground residue C input did not increase CO₂ emissions in croplands due to interaction of cropping system with tillage and N fertilization. Similar findings were observed in grassland. Increased root respiration probably increased CO₂ emission in grassland compared with cropland. Regardless of cropping systems, the emissions were greater with irrigation than without. For reducing CO₂ emission from croplands, no-tilled continuous cropping with recommended rates of N fertilization can be used.

References

- Al-Kaisi MM, Kruse ML, Sawyer JE (2008) Effect of nitrogen fertilizer application on growing season carbon dioxide emission in a corn-soybean rotation. *Journal of Environmental Quality* **37**, 325-332.
- Curtin, D, Wang H, Selles F, McConkey, BG, Campbell CA (2000) Tillage effects on carbon fluxes in continuous wheat and fallow-wheat rotations. *Soil Science Society of America Journal* **64**, 2080-2086.
- Duxbury JM (1994) The significance of agricultural sources of greenhouse gases. *FertilizerResearch* **38**, 151-163.
- Lal R, Kimble, JM, Stewart BA (1995). World soils as a source or sink for radiatively-active gases. In 'Soil Management and Greenhouse Effect. Advances in Soil Science'. (Ed R Lal) pp. 1-8 (CRC Press: Boca Raton, FL).
- Sainju UM, Jabro JD, Stevens WB (2008). Soil carbon dioxide emission and carbon sequestration as influenced by irrigation, tillage, cropping system, and nitrogen fertilization. *Journal of Environmental Quality* **37**, 98-106.

Enzyme activity in a sinkhole undergoing forage renovation

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Abstract

Endophyte-infected tall fescue is a dominant forage grass in the southeastern United States. Efforts are being made to find replacement forages for animal production, but knowledge about such renovation effects on soil biological properties is limited. We compared soil enzyme activity at two depths in three tall fescue cultivars differing in endophyte infection status. The tall fescue was planted in 160 m-long transects spanning a sinkhole, which represents a typical topographic feature in central Kentucky pastures. Soil dehydrogenase and β -glucosidase activity were significantly higher at 0-15 than 15-30 cm depth. At 0-15 cm depth both enzymes had significantly lower activity in the newly established grasses than in the undisturbed control. Soil dehydrogenase activity declined in the order: Undisturbed pasture (control) > KYFa9301 (endophyte-free) = KY 31 (endophyte infected and alkaloid-producing) > KYFa9301/A584 (endophyte-infected, alkaloid-free). β -glucosidase activity declined in the order: Undisturbed pasture (control) = KYFa9301 > KY 31 = KYFa9301/A584. No statistically significant correlation was found between enzyme activity and soil pH, TC, TN, clay content, or moisture content. The effects of forage transition were observable in the short term, and presence or absence of endophyte in tall fescue appears to affect soil biological activity.

Key Words

Soil dehydrogenase, β -glucosidase, tall fescue, *Neotyphodium coenophialum*, karst topography.

Introduction

Tall fescue (*Festuca arundinaceae* Schreb.) is a cool-season forage grass that dominates pastures in the southeastern United States. Kentucky 31 (KY 31) is a dominant cultivar in this region, but unfortunately suffers from infection by the endophyte *Neotyphodium coenophialum*, which produces various ergot alkaloids (e.g. lysergic acid, ergovaline) that are detrimental to animal growth and productivity (Siegel and Bush 1996). Much effort is currently devoted to replacing KY 31 with alternate forage species, including endophyte or alkaloid free tall fescue. Although there has been some research on the effects of such transition in terms of soil C (Franzluebbers *et al.* 1999; Franzluebbers and Hill 2005; Franzluebbers and Stuedmann 2005) little is known about the immediate effects of transition on sensitive soil quality indicators reflecting C dynamics. In central Kentucky, potential effects of forage transition are exacerbated by the underlying karst topography. Numerous sinkholes that develop in this landscape provide rapid conduits for surface contaminants into shallow groundwater resources. Howell *et al.* (1995) have already demonstrated that fecal coliforms from grazing cattle in karst environments can rapidly percolate through soil into springs and streams. Transition to forages more palatable to grazing animals could therefore have unintended detrimental effects with respect to groundwater quality. The objective of this study was to determine if transition from native pasture to one of three tall fescue cultivars with or without endophyte and/or alkaloid production would influence enzyme indicators of soil quality related to C transformations.

Methods

Research site

The research was performed at the University of Kentucky Animal Research Center in Woodford Co., Kentucky, USA, approximately 18 km east of Lexington. The soil at this site is classified as a Maury silt loam (Typic hapludalf) with 6-12% slopes. An existing sinkhole in permanent pasture – a mixture of Kentucky bluegrass (*Poa pratensis* Linn.) and tall fescue – was treated with glyphosate herbicide (RoundupTM) to remove the existing vegetation in Fall 2008, and again in Spring 2009. Three tall fescue cultivars were direct seeded into the killed sod in April 2009:

1. Control – Existing pasture
2. KY 31 – Endophyte infected and alkaloid-producing tall fescue
3. KYFa9301/A584 – Endophyte infected, non alkaloid-producing tall fescue
4. KYFa9301 – Endophyte- and alkaloid-free tall fescue.

The experiment design consisted of four nonreplicated treatments spanning the sinkhole. Each treatment was a strip 160 m-long by 3 m-wide separated by 2 m-buffers. In July 2009, after the new forages were well established, soils were collected at 10-m intervals along four transects representing the three forage treatments and an undisturbed control (Figure 1). Each location was sampled at two depths: 0-15 and 15-30 cm. Soils were manipulated to break up large aggregates, air dried, and stored at 4° C until use. To make treatment comparisons of enzyme activity paired difference *t*-tests assuming equal variance were performed for enzyme activity in each treatment at each sampling location with the null hypothesis of no treatment difference between treatments for the entire population of sample locations. For the purpose of statistical analysis the design was treated as a systematic distribution (Hurlbert 1984).

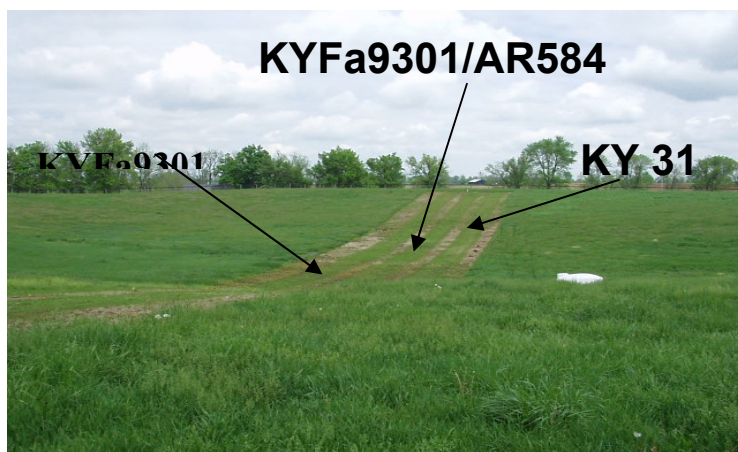


Figure 1. Distribution of forage treatments in the sinkhole site at the University of Kentucky Animal Research Center in Woodford Co.

Soil enzyme activity

Soil dehydrogenase was assessed colorimetrically in triplicate by the method of Tabatabai (1996), which measures the production of TPF (Triphenyltetrazolium formazon) from the reduction of TTC (Triphenyltetrazolium chloride) in samples incubated at 37° C for 24 h. β -glucosidase activity was measured colorimetrically in triplicate by the production of para nitrophenol (PNP) from the hydrolysis of para nitrophenol- β -D-glucopyranoside in samples incubated 1 h at 37° C (Tabatabai 1996). Both sets of soil enzyme activity utilized reagent blanks to correct for background absorbance.

Results

The experiment design and the varied topography precluded a simple comparison of treatment means for each enzyme. To minimize the potential for pseudoreplication we compared differences among treatments at each sampling location. This approach does not preclude a systematic bias across treatment locations at each sampling location, but spatial analysis indicated that within the region encompassed by each sampling location soil properties (pH, total C, total N, clay content) were random and therefore enzyme differences between treatments were ascribed to treatment effects (El-Naggar *et al* 2010). Soil Enzyme Activity Soil dehydrogenase activity was significantly higher ($p < 0.05$) at 0-15 than 15-30 cm in all treatments (Figure 2). Likewise, the dehydrogenase activity in the control was significantly higher ($p < 0.05$) in the control compared to all other treatments. Cultivar KYFa9301/A584, which is infected by a novel non-alkaloid-producing endophyte, had significantly lower soil dehydrogenase activity than the other two treatments.

β -glucosidase activity was significantly higher ($p < 0.01$) at 0-15 cm than 15-30 cm in all treatments. Activity in the control ranged from 147 to 228 $\mu\text{g PNP g}^{-1} \text{h}^{-1}$ at 0-15 cm (Figure 3), which was significantly ($p < 0.05$) higher than fescue treatments containing endophytes for the same depth. There were also significant differences between the fescue treatments at 0-15 cm depth with the general trend for activity being: Undisturbed pasture (control) = KYFa9301 > KY 31 = KYFa9301/A584 (Figure 4).

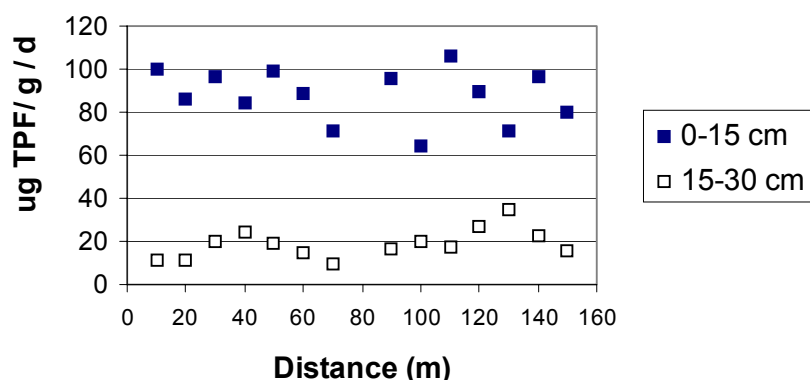


Figure 2. Soil dehydrogenase activity in the undisturbed control at two depths. All other treatments showed the same result, and are omitted for simplicity. The center of the sinkhole appears at approximately 80 m distance.

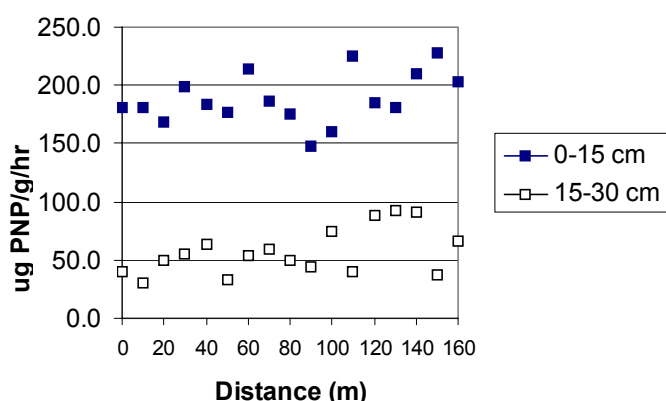


Figure 3. β -glucosidase activity in the undisturbed control sample at two soil depths. All other treatments showed the same result, and are omitted for simplicity. The center of the sinkhole appears at approximately 80 m distance.

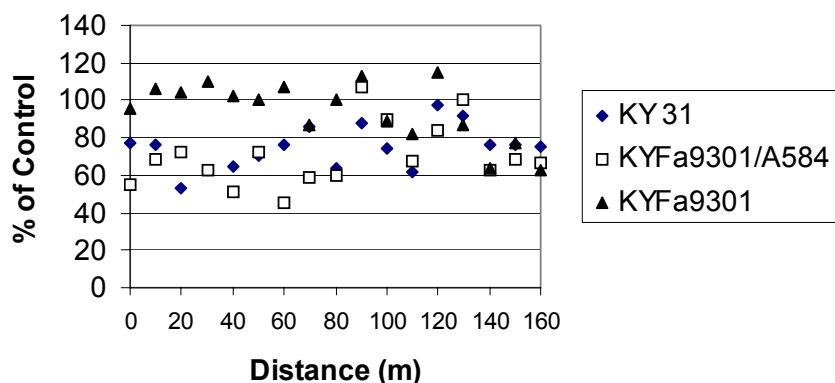


Figure 4. β -glucosidase activity in fescue treatments at 0-15 cm depth relative to the undisturbed control at each position.

Conclusion

Four months after pasture renovation and replacement of the existing sod with alternative fescue cultivars there were clear differences in the level of dehydrogenase and β -glucosidase activity. In each case the transition resulted in decreased activity, which indicates that procedures for pasture renovation that involve killing the existing sod with herbicide before reseeding can have an adverse effect on soil biological activity. There was some evidence that the choice of replacement fescue cultivars had an influence on the subsequent soil enzyme activity. KYFa9301/A584, which is a KY 31 derivative containing a novel endophyte that produces no alkaloid, typically had the lowest soil enzyme activity of any of the cultivars tested. Despite having a symmetrical appearance, the sinkhole was asymmetric with respect to enzyme activity. There was evidence for spatial organization of the enzyme data (El -Naggar *et al* 2010). Most likely, this was because the south-facing slope for the distance 80-160m was both warmer and shallower than the north-facing slope

Acknowledgements

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References

- El-Naggar E, Coyne MS Phillips TD (2010) Spatial variability of soil enzymes in a sinkhole undergoing forage transition. Proceedings of the 19th World Congress of Soil Science, Brisbane. www.19wcss.org.au.
- Franzluebbers AJ, Hill NS (2005) Soil carbon, nitrogen and ergot alkaloids with short-and long-term exposure to endophyte-infected and endophyte-free tall fescue. *Soil Science Society of America Journal* **69**, 404-412.
- Franzluebbers AJ, Nazih N, Stuedmann JA, Fuhrmann JJ, Schomberg HH, Hartel PG (1999) Spatial carbon and nitrogen pools under low- and high- endophyte infected tall fescue. *Soil Science Society of America Journal* **63**, 1687-1694.
- Franzluebbers AJ, Stuedmann JA (2005) Soil carbon and nitrogen pools in response to tall fescue endophyte infection, fertilization, and cultivar. *Soil Science Society of America Journal* **69**, 396-403.
- Howell JM, Coyne MS, PL Cornelius PL (1995) Fecal bacteria in agricultural waters of the bluegrass region of Kentucky. *Journal of Environmental Quality* **24**, 411-419.
- Hurlbert SH (1984) Pseudoreplication and the design of ecological field experiments. *Ecological Monographs* **54**, 187-211.
- Siegel MR, Bush LP (1996) Defensive chemicals in grass-fungal endophyte associations. *Recent Advances in Phytochemistry* **30**, 81-119.
- Tabatabai MA (1996) Soil enzymes In 'Methods of Soil Analysis, part 2: Microbiological and Biochemical Properties'. (Eds RW Weaver *et al.*) pp. 775-833. (Soil Science Society of America, Inc., Madison, WI).

From innovation to adaptation: A 30-year SEE lesson from the evolution of saline-alkali soil management in Manasi River watershed, China

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Abstract

Previous studies on saline-alkali soil management mostly followed an instrumental “prediction and control” approach dominated by technical end-of-pipe solutions. However, those “integrated” instrumental solutions frequently perished due to the growing social & economic uncertainties in financial support, legal insurance, expertise service and other factors. This investigation summarizes the 30-year period of saline-alkali soil management – the social and economic and ecological (SEE) management innovation – its adoption, diffusion, adaptation and transformation in Manasi River watershed of northern Xinjiang. This area was experiencing three distinct SEE management stages over last three decades: preliminary development of instrumental desalination techniques with little support from SEE system from 1978 to 1988; rapid development of integrated instrumental and engineering desalination technique system but still separating from social policy and economic system from 1989 to 2000; and only in the recent decade have successful practices of integrated desalination technique system been achieved following the transformation of local SEE supporting system to collaborate well with instrumental approaches. The results of GIS analysis (Fragatats 3.3) and historical documents provide data evidence for above three transition stages. The total area of saline and alkali land was increased by 32.7%, 47.6% during the first two decades but decreased by 11.9% in the recent decade. The numbers of saline land patches were 116, 129 and 121 in 1989, 2000 and 2007 respectively, a similar trend to the changes of total area. However, both perimeter-area fractal dimension (PAFD) and splitting index (SI) continued to increase, with values of 1.265, 1.272 and 1.279 for PAFD and 259.29, 269.68, 272.92 for SI in 1989, 2000 and 2007, respectively. It suggests that saline and alkaline land distribution had been fragmented, and sequestered into salt micro-catchments within whole oasis ecosystems. This case is largely associated with effective adoption of integrated engineering and biological desalination programs as a result of local SEE saline-alkali soil management innovation.

Key Words

Social & economic & ecological (SEE) system, saline-alkali soil, instrumental solution, innovation, China.

Introduction

‘It seems to me that the nature of true tragedy is when something is so badly broken that with the best will in the world, you can’t put it back together again and what was broken has to stay broken’ (Presentation by Salman Rushdie, 29 August 2005 and cited by Weinstein 2008). Manasi River watershed is located in northern Xinjiang of China and is a typical arid oasis ecosystem. Before 2000-year, saline affected farmland area was up to 47.9% of total farmland area due to inefficient water resource management strategy (cited from local government official document 2008). Unreasonable water resource utilization was a main driving force to induce the formation and aggravation of soil salinity and alkalinity in arid/semiarid oasis ecosystem (Jolly *et al.* 2008). Aggravation of salinization seemed to bring local agricultural production, social-economic development and even human habitation into irreversible ecological crisis. High groundwater level and excessive surface water consumption were the major natural factors causing this crisis. Although some instrumental solutions started to be put into practice from 1990 to 2000, the salinized area still increased from 850 km² to 1254 km² as a result of lacking enough supports and interactions from social-economic and policy systems. Initially, water resources management followed an instrumental “prediction and control” approach dominated by technical end-of-pipe solutions (Pahl-Wostl *et al.* 2008). Relevant efforts were paid on salineland management using instrumental or purely engineering approaches in saline affected area (Luedeling *et al.* 2005). However, this approach no longer works well, because it cannot adequately deal with the growing uncertainties, different stakeholder perspectives and growing interdependence that are characteristic for today’s resource management issues (Pahl-Wostl *et al.* 2008). New approaches can be

found only within the framework of coupled human and natural systems (CHANS). The CHANS has been experiencing unprecedented rapid changes and progressively tighter couplings at multiple scales (Liu *et al.* 2007).

Since 2000, local government issued a series of reclamation and management policies and measures including legal regulations of water allocation and water quality monitoring, administrative policies of irrigation engineering and technical program, economic regulations on land use and management, social organizations of community participative water management association and scientific technical standards of farmland drainage and irrigation, etc. The updated management system enabled pure instrumental and engineering solutions to a successful saline & alkali soil reclamation practice. Putting the knowledge gained from such integrated studies into social, economic and environmental (SEE) decision-making processes is essential for achieving productive and sustainable CHANS (Liu *et al.* 2007). The objective of this study is to conclude a successful practical case about integrated SEE technique to optimize water utilization system, restrain saline land enlargement and obtain sustainable human-nature relationship.

Methods

Study sites

Manasi River watershed is located at the north Xinjiang of China (85°01' - 86°32'E, 43°27' - 45°21'N) with the total area of $2.43 \times 10^4 \text{ km}^2$, geographically originating from the northern Tianshan Mountain and mainly covering Shihezi city, Shawan county and Manasi County of Xinjiang (Figure 1). It is adjacent to the southern Gurbantunggut Desert and belongs to typical arid oasis ecosystem with 110-200 mm annual precipitation. Historically, this area was a nomadism oasis which started to be exploited as agricultural reclamation area since 1955. The total population increased from about 50,000 at that time to a current population of 910,000. Agricultural production is the main resource of local GDP and the water consumption of agriculture accounts for 90% of total water resource use.

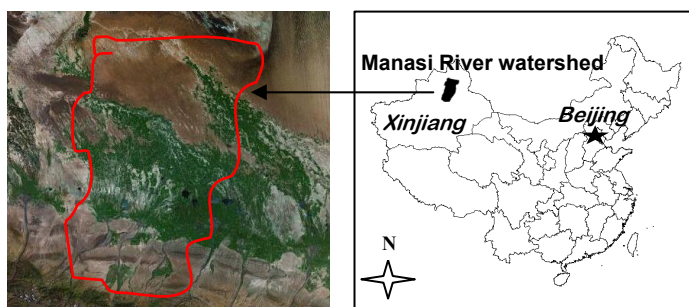


Figure 1. Manasi River watershed sketch map.

Data collection

Three periods of TM, ETM and TM satellite images with 30 m of spatial resolution in 1989, 1999, and 2007 were used to evaluate the dynamics of saline & alkaline soil distribution in Manasi River watershed. Local historical documents concerning farmland area survey were also used as accessorial data. Social, economic and ecological (SEE) developmental data were harvested from local historical documents.

Analysis method

Spatial data are extracted from satellite images over last three decades mentioned above. The landscape analysis program Fragatats 3.3 was employed to analyze the landscape indices, *CA* (Class Area, km^2), *NP* (Number of Patches), and *PAFRAC* (Perimeter-Area Fractal Dimension). *SPLIT* (Splitting Index) was presented to concentrate and scatter characters of land patches.

Results

The evolution of social-economic strategies in preventing the enlargement of saline land over three decades
Manasi watershed's social, economic and environmental (SEE) strategies include economic system, social system and instrumental and engineering system at four levels such as individual farmer, community level, sub-watershed level and whole watershed. In past three decades this area has been the fastest growing among major cities in Xinjiang, with an almost 12% annual increase domestic product (GDP). The instrumental and engineering technique service system in this area has ranged from expertise system, information centre, on-site technical service and community organization (Figure 2). It directly interacted

with and served for 115,000 individual farmers in cropping management and fertilizer application including dripping irrigation introduction and soil salt removal and attenuation techniques. Beyond the individual farmer level, local government had conducted a series of soil salt reduction programs including engineering salt-removing system (drainage network to lower the groundwater line by pipe system among over 85% croplands, leakage-preventing trench system and land level-up program throughout all the fields) and biological desalinization projects (salt-washing system by rice cropping and salt-accumulating system by planting trees along with irrigation trenches throughout the watershed) at sub-watershed level. Importantly, all the solutions mentioned rested on a preliminary trial stage and very little was put into practice before 1989, and only till after 1990 had they been developed into a relatively integrated instrumental and engineering technique system. It was since 2000 that overall SEE systems had been built up well, which had coupled interactions with instrumental technique system, including funding input, market system, industrial pattern upgrade, administrative policy, legal insurance system and environmental education (Figure 2).

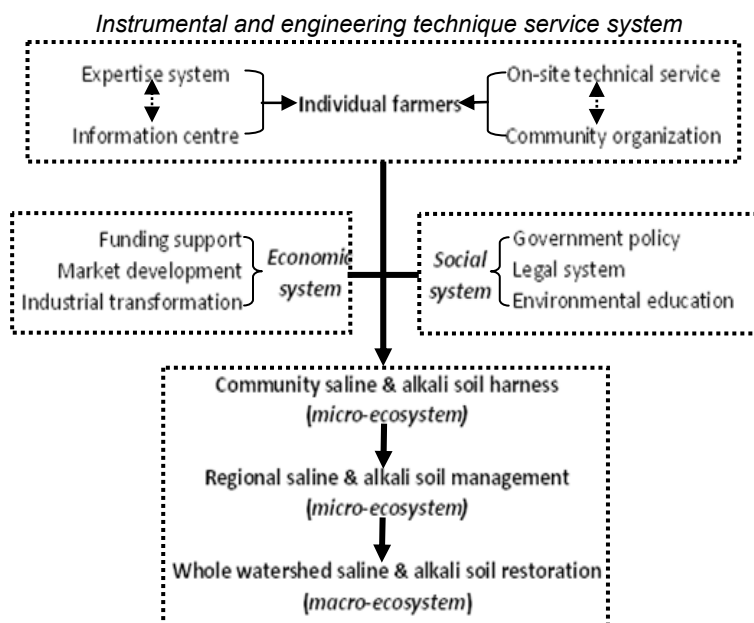


Figure 2. Social, economic & ecological (SEE) system for saline & alkaline soil management at Manasi watershed.

The temporal-spatial dynamics of saline & alkaline soil distribution over three decades

Saline and alkaline soil distribution varied from periods and regions (Figure 3). Total area and patch number were 850.2, 1254.6, 1105.4 km² and 116, 129 and 121 in 1989, 2000 and 2007 respectively, showing a dynamics of “first increase and then decrease” trend. However, perimeter-area fractal dimension (PAFD) and splitting index (SI) remained increasing, being up to 1.265, 1.272 and 1.279 for PAFD and 259.29, 269.68, 272.92 for SI in 1989, 2000 and 2007 respectively (Figure 4), suggesting that the spatial pattern of salinized land tended to be shrunk and sequestered into micro-catchment patches .

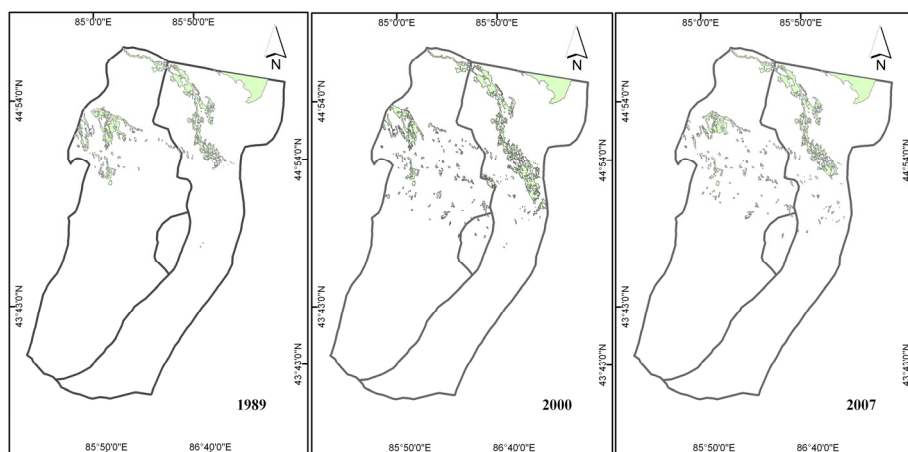


Figure. 3 The temporal & spatial distribution of saline land at Manasi watershed from 1989 to 2007.

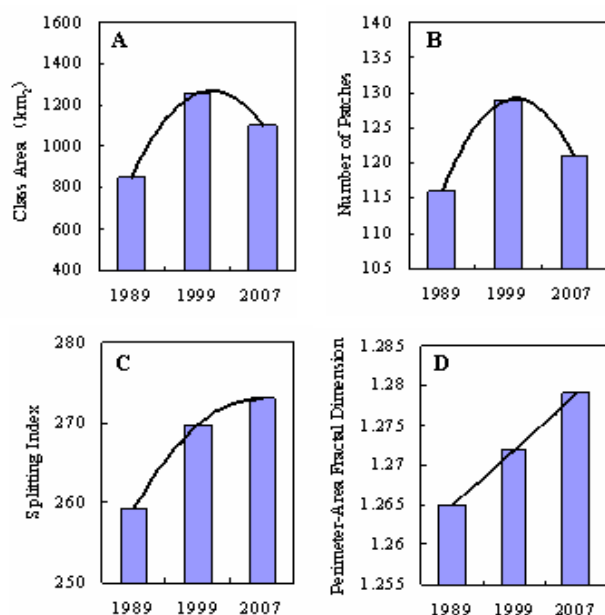


Figure 4. Landscape parameters dynamics of salineland distribution at Manasi watershed from 1989 to 2007.

Conclusion

The lack of SEE harness program will lead to the invalidation of instrumental solutions in practice of saline land control and ecological restoration. Also, the inadequate combination of SEE program into pure instrumental and engineering technique system will let land the salinization crisis get out of control, while the pure technical solution is developed to a “so-called” integrated highly efficient system. Only in fine combination between SEE system and instrumental integrated system is the declamation and utilization of saline and alkaline land able to be gradually sustainable and environment-friendly. Thirty-year lesson from the evolution of saline land at Mansi River watershed provides a promising practical case for other similar areas worldwide. As proposed by Liu *et al.* (2007), coupled human and natural system (CHANS) cannot work well except where the SEE factors are taken into consideration.

Acknowledgement

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References

- Pahl-Wostl C, Mostert E, Tabara D (2008) The Growing Importance of Social Learning in Water Resources Management and Sustainability Science. *Ecology and Society* **13**, 24.
- Jolly ID, McEwan KL, Holland KL (2008) A review of groundwater-surface water interaction in arid/semiarid wetlands and the consequences of salinity for wetland ecology. *Ecohydrology* **1**, 43-58.
- Luedeling E, Nagieb M, Wichern F, Brandt M, Deurer M, Buerkert A (2005) Drainage, salt leaching and physico-chemical properties of irrigated man-made terrace soils in a mountain oasis of northern Oman. *Geoderma* **125**, 273-285.
- Wals AEJ (2007) ‘Social Learning Towards a Sustainable World’. (Wageningen Academic Publishers: The Netherlands).
- Weinstein MP (2008) Ecological restoration and estuarine management: placing people in the coastal landscape. *Journal of Applied Ecology* **45**, 296-304.
- Liu JG, Dietz T, Carpenter SR, Folke C, Albert M, Redman CL, Schneider SH, Pell AN, Lubchenco J, Taylor WW, Ouyang ZY, Deadman P, Kratz T, Provencher W (2007) Coupled human and natural systems. *Ambio* **36**, 639-648.
- Zhu HM (1999) Harness the Saline – alkali soil in Xinjiang. *Journal of Shihezi University (Natural Science)* **3**, Sup. 78-82.
- Program of Reclamation and Utilization of Saline and Alkaline soil at Regional Level (2008) ‘Local Government Official Program’. (Developmental and Reforming Committee of Xinjiang Bintuan).

Impacts of long-term no-tillage and conventional tillage management of spring wheat-lentil cropping systems in dryland Eastern Montana, USA, on fungi associated to soil aggregation

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Abstract

Lentil (*Lens culinaris* Medikus CV. Indianhead) used to replace fallow in spring-wheat (*Triticum aestivum*) rotation in the semi-arid Eastern Montana USA, may improve soil quality. We evaluate the 14 years influence of continuous wheat under no-tillage (WNT), fallow-wheat under conventional tillage (FCT) and no-tillage (FNT), lentil-wheat under tillage (LCT) and no-tillage (LNT) on soil formation and stability, and on the amount of immunoreactive easily-extractable glomalin (IREEG) and soil aggregating basidiomycete fungi in the 4.75-2.00, 2.00-1.00, 1.00-0.50, 0.50-0.25, and 0.25-0.00 mm aggregate-size classes, at 0-5 cm soil depth. The 4.75-2.00 mm aggregate proportion was higher in LNT than FNT and higher in LT than FT treatments and mean weight diameter (MWD) was higher when lentil was used to replace fallow under NT. No-till systems had higher glomalin and basidiomycete amount than CT in all aggregate-size classes and glomalin was higher in LNT than FNT in aggregate-size classes less than 0.50 mm. We conclude that residue input in NT systems triggers fungal populations which are involved in soil binding in aggregates, and that replacing fallow by lentil in spring wheat rotation in dryland seems to favor aggregate formation/stability under NT probably by increasing N fertility during the course of 14 years.

Key Words

Tillage, soil aggregation, lentil, fallow, spring wheat, glomalin, basidiomycete fungi.

Introduction

Wheat-fallow rotations commonly practiced in the semiarid Northern Great Plains of the United States, resulted in substantial soil erosion, deterioration of the quantity and quality of soil organic matter, and increased greenhouse gas emissions (Sainju *et al.* 2008; 2009). By using a legume to replace fallow in a wheat-fallow rotation, there has been a gradual reduction in fertilizer-N requirement after about 6 years (Campbell *et al.* 1992). Legume-based cropping systems will not only reduce nitrogen losses, but they may also increase the proportion of crop residue carbon that is sequestered in stable soil organic matter (Drinkwater *et al.* 1998). It appears that rotations containing grain legumes will also result in greater microbial activity (Biederbeck *et al.* 2005). However, the effects of legume cropping in dryland spring wheat rotations on soil aggregation and fungal population involved in soil binding is not known. Our objective was to study aggregate formation and stability, and the populations of fungi associated to soil aggregation in different aggregate-size classes in lentil spring wheat systems in comparison to fallow spring wheat and continuous spring wheat as influenced by tillage.

Methods

The study was located in Culbertson, Montana, USA on a Williams loam (fine-loamy, mixed Typic Argiboroll). Annual precipitation is 340 mm. The experimental design is detailed in Pikul *et al.* (1997). Treatments consist of two tillage systems (conventional tillage, CT, and no-tillage, NT) and five cropping systems (no-tilled continuous wheat, CWNT, no-tilled wheat-fallow, FNT, conventional tilled wheat-fallow, FCT, no-tilled wheat-lentil, LNT, conventional tilled wheat-lentil, LCT) as split-plot treatments in a randomized block design with four replications. Indianhead lentil (*Lens culinaris* Medikus CV. Indianhead) was grown as green manure (L) and terminated either mechanically (LCT) or by herbicides (LNT) using a mixture of glyphosate (2.3 L/ha) and 2,4-D (0.54 kg/ha). Indianhead lentil was used in this study because this legume has intermediate topgrowth N yield and the seed was available at low cost. Soil samples collected to a depth of 5 cm were processed within 24-48 h for determinations of soil aggregation. Aggregates were separated by dry sieving and the aggregate proportion (g aggregate/kg soil) was measured in 4.75-2.00, 2.00-

1.00, 1.00-0.50, 0.50-0.25, and 0.25-0.00 mm aggregate-size classes. Mean weight diameter (MWD), used as an index of aggregate stability, was calculated according to the procedure described by Kemper and Rosenau (1996). The immunoreactive fraction of the easily extractable glomalin (IREEG) produced by arbuscular mycorrhizal fungal hyphae was determined by methods previously described (Wright and Upadhyaya, 1998). An enzyme-linked immunosorbent assay was used to quantify the amount of soil aggregating basidiomycete fungi (Caesar-TonThat *et al.* 2001).

Results and discussion

The aggregate proportion and MWD significantly differed among treatments at 0-5 cm soil depth (Table 1). No-tilled continuous spring-wheat (WNT) and wheat-lentil (LNT) treatments had the highest proportion of 4.75-2.00 mm aggregate-size class compared to the other treatments with a subsequent decrease in the 0.25-0.00 mm aggregate-size class. Regardless of tillage, wheat-lentil treatments had higher proportions of 4.75-2.00 mm aggregate-size class compared to wheat-fallow treatments. Mean-weight diameter (MWD) was higher in LNT than in FNT and FT, and was the highest in WNT and the lowest in FCT among the treatments. These results suggest that continuous spring-wheat and wheat-lentil treatments under NT practice are more suitable in dryland Montana USA than wheat-fallow treatments in improving aggregate formation and stability. The increase of aggregate stability in LNT compared to FNT could be attributed to lentil incorporation of readily decomposable and N-rich substrate which provides a better environment for microbial growth and activities that are involved in the production of soil binding agents. These data are consistent with Biederbeck *et al.* (1998) who had found an increase in soil aggregation in lentil system compared to wheat fallow systems at the top soil surface layer (0-10 cm) in the semiarid climate of the Canadian prairie.

Table 1. Effects of tillage and cropping systems on the proportion of soil aggregates and mean weight diameter (MWD) at 0-5 cm soil depth.

Treatments*	Aggregate proportion in size class (g aggregate/kg soil)					MWD*** (mm)
	4.75-2.00 mm	2.00-1.00 mm	1.00-0.50 mm	0.50-0.25 mm	0.25-0.00 mm	
WNT	394.21a**	134.07a	106.29ab	135.90b	215.60c	0.51a
FCT	96.94c	94.67b	109.77ab	203.48a	495.30a	0.39c
FNT	188.06b	110.09ab	96.27b	205.27a	411.44b	0.43c
LCT	151.14b	104.70ab	132.63a	205.19a	406.67b	0.44bc
LNT	354.69a	111.02ab	104.87ab	189.12ab	240.31c	0.48ab
Means						
NT	308.85a	118.48a	102.48a	181.40a	289.12b	0.48a
CT	124.042b	99.65b	121.20b	204.33a	450.99a	0.42b

*Tillage and cropping systems are WNT, no-tilled continuous wheat, FCT, conventional tilled wheat-fallow, FNT, no-tilled wheat-fallow, LCT, conventional tilled wheat-lentil, and LNT, no-tilled wheat-lentil.

**Numbers followed by different letters within a column are significantly different at $P \leq 0.05$ by the Honestly Significant

Difference procedure of Tukey-Kramer.

***MWD, mean weight diameter.

Treatments had significantly different concentrations of immunoreactive easily extractable glomalin (IREEG) and soil aggregating basidiomycete fungi in the different aggregate-size classes (4.75-2.00, 2.00-1.00, 1.00-0.50, and 0.50-0.25 mm) (Figures 1 and 2). Figure 1 indicates that IREEG was higher in all NT treatments compared to CT in all aggregate-size classes. Continuous spring wheat under NT (WNT) had the highest concentration of IREEG compared to the other treatments in aggregate-size classes (4.75-2.00, 2.00-1.00, and 1.00-0.50 mm) but was lower than LNT in the 0.50-0.25 mm aggregate-size class. There was significant higher glomalin concentration in aggregate-size classes 1.00-0.50 and 0.50-0.25 mm in LNT than FNT. The quantification of the soil aggregating basidiomycete fungi indicated that in general WNT had the largest amount of basidiomycetes than the other treatments (Fig. 2). There was no difference in the amount of fungi between FNT and LNT and between FCT and LCT, but FNT and LNT were significantly higher than FCT and LCT. The data indicated that regardless of cropping systems, NT in the Northern Plains dryland favor the concentrations of IREEG and soil aggregating basidiomycete fungi on the top soil surface (0-5 cm) because mycorrhizal and basidiomycete fungi are known to be sensitive to soil disturbance caused by tillage (Caesar-TonThat *et al.* 2001; Wright and Upadhyaya, 1998), and plant residue input resulted from

NT systems favor the survival and growth of these fungi. Results showing a higher amount of IREEG in LNT compared to FNT are consistent with Doubs *et al.* (1997) who have reported that that organic matter increased AM fungal hyphae growth.

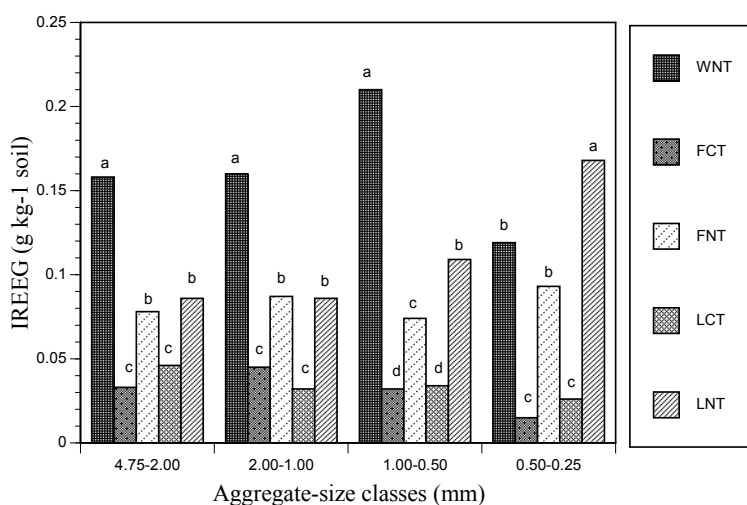


Figure 1. Concentrations of immunoreactive easily extractable glomalin (IREEG) at 0-5 cm soil depth. Tillage and cropping systems are no tilled continuous spring wheat (WNT), conventional tilled wheat-fallow (FCT), no-tilled wheat-fallow (FNT), conventional tilled wheat-lentil (LCT), and no-tilled wheat-lentil (LNT). Bars followed by the same lowercase letters are not significantly different at $P \leq 0.05$.

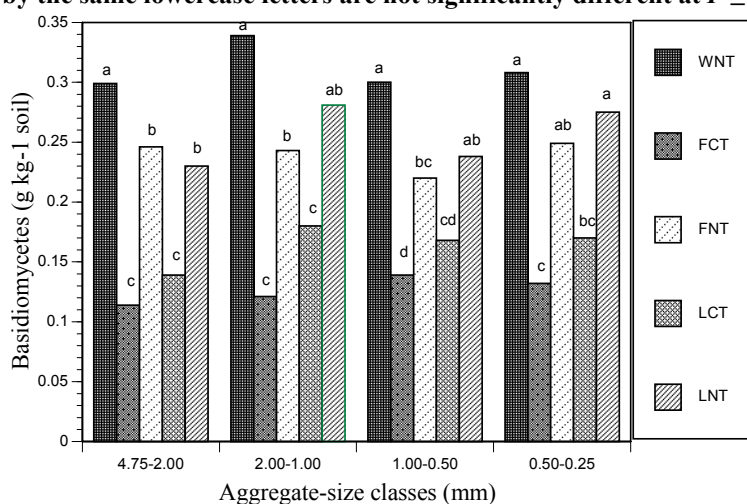


Figure 2. Amount of soil aggregating basidiomycete fungi at 0-5 cm soil depth as detected by ELISA. Tillage and cropping systems are no tilled continuous spring wheat (WNT), conventional tilled wheat-fallow (FCT), no-tilled wheat-fallow (FNT), conventional tilled wheat-lentil (LCT), and no-tilled wheat-lentil (LNT). Bars followed by the same lowercase letters are not significantly different at $P \leq 0.05$.

Conclusion

The use of lentil to replace fallow in spring wheat rotation system improved soil aggregate formation regardless of tillage management and aggregate stability was improved in LNT compared to FNT. Under NT, wheat-lentil rotation improved glomalin concentration when compared to wheat-fallow rotation. Regardless of cropping systems, aggregate stability, glomalin (IREEG) concentrations and basidiomycete fungi were higher under NT than CT.

References

- Beiderbeck VO, Campbell CA, Rasiah V, Zentner RP, Wen G (1998) Soil quality attributes as influenced by annual legumes used as green manure, *Soil Biol. Biochem.* **30**, 1177-1185.
- Beiderbeck VO, Zentner RP, Campbell CA (2005) Soil microbial populations and activities as influenced by legume green fallow in a semiarid climate, *Soil Biol. Biochem.* **37**, 1775-1784.
- Caesar-TonThat TC, Shelper WL, Thorn RG, Cochran VL (2001) Generation of antibodies for soil-aggregating basidiomycete detection to determine soil quality, *Appl. Soil Ecol.* **18**, 99-116.

- Campbell CA, Zentner RP, Selles F, Biederbeck VO, Leyshon AJ (1992) Comparative effects of grain lentil-wheat and monoculture wheat on crop production, N economy and N fertility in a Brown Chernozem, *Can. J. Plant Sci.* **72**, 1091-1107.
- Doubs DD, Galvez Jr L, Franke-Snyder M, Reider C, Drinkwater LE (1997) Effect of compost addition and crop rotation point upon VAM fungi, *Agric. Ecosyst. Environ.* **65**, 257-266.
- Drinkwater LE, Wagoner P, Sarrantonio M (1998) Legume-based cropping systems have reduced carbon and nitrogen losses, *Nature* **396**, 262-265.
- Kemper WD, Rosenau RC (1986) Aggregate stability and size distribution, in A. Klute A (Ed.), *Methods of soil analysis: physical and mineralogical methods*, (Part 1), 2nd ed., ASA, SSSA Spec Publ No 9, Madison, WI, 1986, pp. 425-442.
- Pikul Jr JL, Aase J, Cochran VL (1997) Lentil green manure as fallow replacement in the semiarid Northern Great Plains, *Agron. J.* **89**, 867-874.
- Sainju UM, Jabro JD, Stevens WB (2008) Soil carbon dioxide emission and carbon content as affected by irrigation, tillage, cropping system, and nitrogen fertilization, *J. Environ. Qual.* **37**, 98-106.
- Sainju UM, Caesar-TonThat TC, Jabro JD (2009) Carbon and nitrogen fractions in dryland soil aggregates affected by long-term tillage and cropping sequence, *Soil Sci. Soc. Am. J.* **73**, 1488-1495.
- Wright SF, Upadhyaya A (1998) A survey of soils for aggregate stability and glomalin, a glycoprotein produced by hyphae of arbuscular mycorrhizal fungi, *Plant Soil* **198**, 97-107.

Influence of earthworms on the growth of cotton and wheat plants in contrasting soil types

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Abstract

The development and survival of two exotic species of earthworms, *Aporrectodea caliginosa* and *A. longa* (Lumbricidae) were measured in two soils, a grey clay commonly used for cotton production in eastern Australia and a sandy loam available at a commercial garden centre. Most earthworms survived in all treatments, indeed there was evidence of reproduction of *A. caliginosa*, but the growth of both earthworm species was generally less in the clay soil (which had added fertiliser) than in the sandy loam (without fertiliser). Earthworms did not significantly influence the growth of cotton seedlings in either soil type, nor that of the wheat in the clay soil. In the sandy loam however, earthworms positively influenced growth. Overall, the results suggest that earthworms do not influence cotton's early vegetative growth to the same extent as they do that of wheat, and their influences on plant growth can vary with soil type.

Key Words

Earthworms, cotton, wheat, plant production, glasshouse.

Introduction

Earthworms, sometimes referred to as “ecosystem engineers” are well known for influencing soil structure, fertility and plant production in various agricultural ecosystems (Edwards and Bohlen 1996). The benefits earthworms bring include improved aggregate stability, increased porosity, aeration and water infiltration, enhanced nutrient availability, retention of nutrients on-farm (through efficient burial of surface organic matter and fertilisers and prevention of leaching in surface water flow), deeper rooting of plants, and reductions in the incidence of root diseases. Several of these benefits have been demonstrated in Australia, in pasture and cereal cropping systems (Baker *et al.* 2003; Baker 2004). In contrast, very little is known of the biology and functional role of soil macro-fauna such as earthworms under cotton crops in Australia (and elsewhere). Cotton farming in Australia has traditionally presented several potential hazards for soil macro-fauna (e.g. heavy pesticide use, tillage, flood irrigation) and the abundance of such soil animals has generally been considered rare. However, recent trends in the industry such as reduced (and softer) pesticide use, less tillage and retention of organic matter would seem likely to have opened opportunities for population growth and (re)colonisation by soil fauna such as earthworms.

This work tested the potential for earthworms to influence the growth of cotton in a clay soil commonly used for cotton production in Australia. In the absence of rigorous knowledge of the fauna in cotton fields, two species of earthworm, commonly found elsewhere in Australia, were used: an endogeic species, *Aporrectodea caliginosa* (Savigny), and an anecic species, *A. longa* (Ude) (Lumbricidae). Whether or not these two species could survive in the field in cotton growing regions in Australia is unknown. Previous work (Baker 1998; Baker and Whitby 2003) has shown that lumbricid earthworms can survive well and enhance plant growth in sandy loam soils. We therefore included a sandy loam soil and wheat in the work reported here, by way of comparison.

Methods

Earthworm collection

Both *A. caliginosa* and *A. longa* were collected in late winter from a sandy loam soil beneath a pasture in north-western Tasmania (Cape Grim) where they are abundant. The earthworms were used for the experiment within a few days after returning to Canberra with them from Tasmania. They were transported in moist sphagnum moss and were maintained in a constant temperature cabinet (15°C; 12:12 light dark regime) in moist, commercial, sandy loam prior to experimental use.

Earthworm influences on cotton and wheat growth

Glasshouse experiments were conducted in Canberra, using a soil from near Narrabri, New South Wales used for cotton and wheat production (grey clay, Vertisol) and a commercial sandy loam (“Gardener’s

Choice”, from Garden World, Hume, A.C.T.). Plastic flower pots (approx 30 cm diameter at the top rim x 30 cm tall, free-draining with a drainage hole covered by fine mesh) were filled with soil (9 kg dry soil/pot). The pots were weighed regularly and the soils watered to 20-25% gravimetric soil moisture content (suitable for earthworm survival and growth). There were 10 replicate pots per treatment. The pots were maintained in a naturally lit, cooled (air conditioners on at 25°C) but not heated glasshouse in which air temperatures fluctuated between approximately 0-30 °C during the experiment. The experiment ran during late winter to mid spring. Pots were arranged at random in the glasshouse. The two soils differed in various characteristics [e.g. clay and sandy loam respectively: Clay (% w/w) 51.0 and 8.5; pH (CaCl₂) 7.5 and 6.9; organic matter (% w/w) 1.9 and 6.5; Olsen P (mg/kg) 14 and 29. Total N (% w/w) was the same for both soils: 0.07].

Earthworms (large juveniles or adults) were washed in water, blotted dry, weighed fresh and then added to the pots requiring them. The period in moss after collection voided the earthworm's guts and enabled more exact body mass measurements. The weights of the worms were measured in groups of 15 individuals of the same species. Groups were selected such that their biomass varied little. A group of 15 earthworms of each species was added to each pot and the tops of the pots were “sealed” for 3 days with fine mesh to prevent earthworm escape, whilst they settled in. This density of the earthworms is common at the site of collection in Tasmania. Any earthworms that did not burrow into the soil within a few hours were replaced.

Wheat (*Triticum aestivum* var. Janz) and cotton (*Gossypium hirsutum* L. var. Sicot 189 QA) seeds were sown into the pots a week after the earthworms were added. Either 12 wheat or 12 cotton seeds were planted per pot. These were later thinned to 3 seedlings per pot. Fertiliser (5 g of urea) and 0.05 g ZnSO₄ was added to each pot, where cotton was to be grown, at time of seeding. Whilst the wheat germinated well, the cotton did not (probably due to cool prevailing temperatures). The cotton seedlings that did emerge were of poor quality. The seedling cotton plants were therefore removed and the same pots were reseeded 7 weeks after the addition of the earthworms. Plants were harvested at 9 weeks from sowing for wheat and 7 weeks for cotton (from second sowing). Plants were cut off at soil level, dried at 60°C for 20 hours and weighed. The soil within each pot was hand-sorted for earthworms at harvest and these were washed in water, maintained on moist filter paper overnight (to again void their gut contents) and then weighed fresh.

The 8 treatments thus were : 1. Clay, no fertiliser, wheat and earthworms, 2. Clay, no fertiliser, wheat and no earthworms, 3. Clay, fertiliser, cotton and earthworms, 4. Clay, fertiliser, cotton and no earthworms, 5. S. Loam, no fertiliser, wheat and earthworms, 6. S. Loam, no fertiliser, wheat and no earthworms, 7. S. Loam, fertiliser, cotton and earthworms, 8. S. Loam, fertiliser, cotton and no earthworms.

Results

Earthworm survival in different soils

The earthworms in the Narrabri clay were distributed evenly throughout the soil when it was hand-sorted. The soil was riddled with macropores, indicating substantial earthworm activity. The earthworms in the commercial sandy loam were mostly confined to the top half of the soil, although macropores were observed lower down. The soil in the lower half of the pots was noticeably much wetter than the top half (much more so than in the Narrabri clay treatment), presumably indicating greater drainage in the sandy loam. Whilst the earthworms from the sandy loam were highly active when washed in water at the end of the experiment, they were less so when from the Narrabri clay.

Most earthworms probably survived throughout the experiment. In several instances, more large *A. caliginosa* were found during hand-sorting of the soils in the pots than were initially added to them (n = 15/pot) (Figure 1a), indicating recruitment which masked any mortality that might have occurred. Small juvenile *Aporrectodea* spp. (most likely *A. caliginosa*, but conclusive identifications were not always possible) and cocoons were found in addition to the larger earthworms in several pots, further confirming that reproduction had occurred. More, large *A. caliginosa* were recovered from the pots with sandy loam and cotton, than from the other pots (F = 9.47, p < 0.05). Most recruitment of small earthworms occurred in the pots with sandy loam and cotton. [Note : the pots with cotton growing in them were maintained for a few more weeks than those with wheat – thus there was more time in the latter for recruitment of young]. There were no significant differences between treatments for the numbers of *A. longa* recovered from the pots (F = 1.71, p > 0.05). The biomass/pot for both *A. caliginosa* and *A. longa* decreased during the experiments (Figure 1b). For both species, biomass was greatest in the sandy loam soil and least in the Narrabri clay, at the end of the experiment (F = 4.04, p < 0.05 for *A. caliginosa*; F = 11.10, p < 0.05 for *A. longa*).

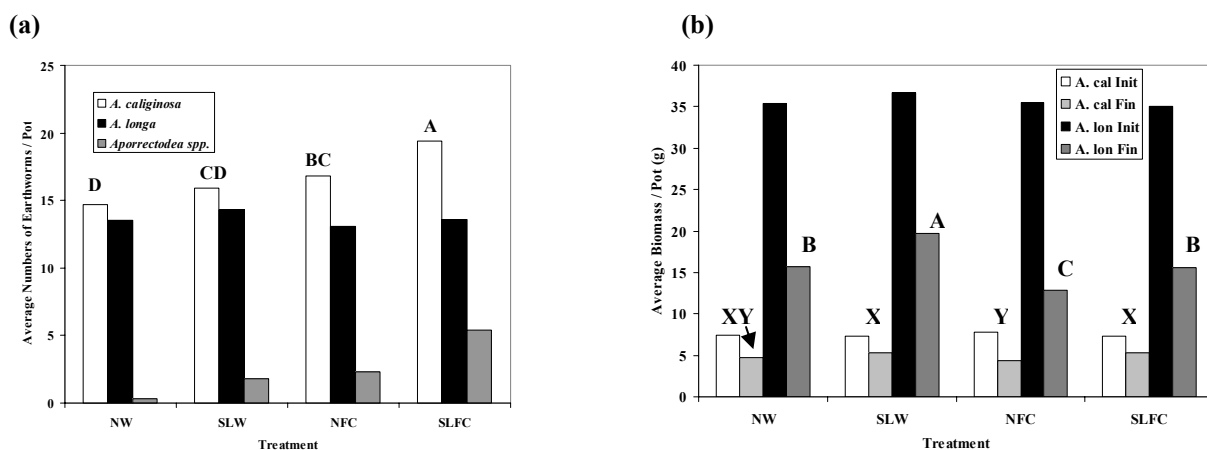


Figure 1. Average numbers (a) and biomass (initial and final) (b) of large *A. caliginosa* and *A. longa* and small *Aporrectodea* spp. after growing wheat (W) or cotton (C) in sandy loam (SL) or clay soil (N) with (F) or without fertiliser. Different letters above the bars indicate significant differences where they occurred between treatments. Small *Aporrectodea* spp. pooled with *A. caliginosa* in (b).

Earthworm influences on cotton and wheat growth

The dry biomass of wheat plants harvested from the pots varied between treatments (Kruskal-Wallis $H = 24.84$, $p < 0.05$), with greatest biomass in the treatment with sandy loam and earthworms and least in the treatments with sandy loam or Narrabri clay and no earthworms (Figure 2a). Most notably, earthworms appeared to increase wheat biomass in sandy loam, but whilst there was a trend in this direction in Narrabri clay it was not significant. The dry biomass of cotton plants harvested from the pots also varied between treatments ($H = 17.55$, $p < 0.05$), with greatest biomass in the treatment with sandy loam and no earthworms and least in the treatment with Narrabri clay and earthworms (Figure 2b).

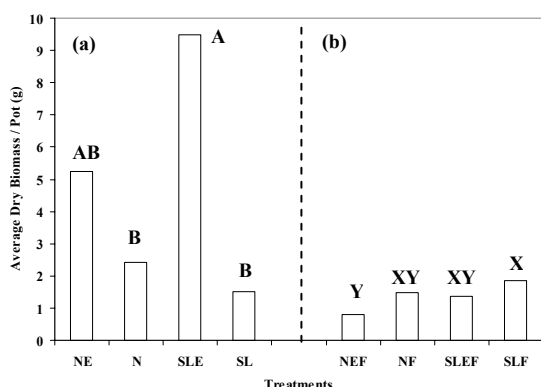


Figure 2. Average dry biomass of wheat (a) and cotton (b) plants growing in sandy loam (SL) or clay soil (N), with or without earthworms (E) and fertiliser (F). Different letters above the bars indicate significant differences between treatments.

Conclusion

These data suggest that earthworms do not enhance early cotton growth. They may of course influence the growth of older plants, but this has yet to be tested. If anything there was a tendency for cotton to grow less in the presence of earthworms. If so, what might be the mechanism? One possibility could be root pruning by earthworms, especially if they are present in high numbers. This has been observed by Cortez and Bouché (1992) for ryegrass in laboratory cultures. In contrast, wheat did respond positively to the presence of earthworms, at least in the sandy loam. It is known that growth responses to earthworms are variable amongst plant species. For example, whilst wheat and oats grew and yielded more in field cages in South Australia which included the lumbricid earthworm, *A. trapezoides*, lupins did not (Baker *et al.* 2003). *A. trapezoides* is very closely related to the *A. caliginosa* we used in this study. Brown *et al.* (1999) also suggested that whilst grain biomass of sorghum and maize can be greatly increased by the addition of earthworms in tropical situations, yields of cowpea and peanuts are reduced. Collectively, these studies suggest that maintenance of high earthworm populations in agricultural soils is not necessarily positive in terms of plant production, and care is called for in terms of using earthworms as indicators of soil “health”.

Cotton was grown in this study with fertiliser, whereas the wheat was not. Whilst the variation in fertiliser influenced the growth of the two plant species, it did not influence the impact earthworms had on the plants (G Baker, unpublished data).

This work was a preliminary foray into the importance of one component of the soil biota, earthworms, on cotton production. We used the exotic *A. longa* and *A. caliginosa* as “model” species, involving two very different burrowing and feeding behaviours, to explore potential influences of earthworms on the growth of cotton and wheat, in particular their early growth. The earthworms survived well in the clay soil from Narrabri, but overall didn’t respond (in terms of weight gain) as well as in the commercial sandy loam. We now need to survey the various cotton growing regions in Australia to determine the earthworm fauna there, evaluate the influences of the most common species on soil properties and cotton production, and consider the merit in increasing the abundance and diversity of such soil communities.

Acknowledgments

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References

- Baker GH (1998) Recognising and responding to the influences of agriculture and other land - use practices on soil fauna in Australia. *Applied Soil Ecology* **9**, 303-310.
- Baker GH (2004) Managing earthworms as a resource in Australian pastures. In ‘Earthworm Ecology’. 2nd Edition (Ed CA Edwards) pp. 263-286. (CRC Press: Boca Raton, USA).
- Baker GH, Amato M, Ladd J (2003) Influences of *Aporrectodea trapezoides* and *A. rosea* (Lumbricidae) on the uptake of nitrogen and yield of oats (*Avena fatua*) and lupins (*Lupinus angustifolius*). *Pedobiologia* **47**, 857-862.
- Baker GH, Whitby WA (2003) Soil pH preferences and the influences of soil type and temperature on the survival and growth of *Aporrectodea longa* (Lumbricidae). *Pedobiologia* **47**, 745-753.
- Brown GG, Pashanasi B, Villenave C, Patron JC, Senapati B, Giri S, Barois I, Lavelle P, Blanchar E, Blakemore RJ, Spain AV, Boyer J (1999) Effects of earthworms on plant production in the tropics. In ‘Earthworm Management in Tropical Agroecosystems’. (Eds P Lavelle, L Brussaard, P Hendrix) pp. 87-147. (CAB International: Wallingford, UK).
- Cortez J, Bouché MB (1992) Do earthworms eat living roots? *Soil Biology and Biochemistry* **24**, 913-915.
- Edwards CA, Bohlen PJ (1996) ‘Biology of Earthworms’ 3rd Edit. (Chapman and Hall: London).

Modelling the provision of ecosystem services from soil natural capital

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Abstract:

Soil ecosystem services and natural capital are often unheralded and generally not well understood. This paper draws on our scientific understanding of soil forming processes, soil classification and functioning and on current thinking about ecosystem services to develop a framework for classifying and quantifying soil natural capital and ecosystem services. The framework consists of five main inter-connected components: (1) Natural Capital, characterised by standard ‘soil properties’; (2) Natural Capital Formation, Maintenance and Degradation Processes; (3) Provisioning, Regulation and Cultural Ecosystem Services; (4) Drivers (Anthropogenic and Natural) of soil processes and related services; (5) Human Needs fulfilled by soils’ services. We then show how such framework is being used to build a model of the provision of soils’ ecosystem services on New Zealand dairy farms.

Key Words

Soil properties, processes, capital formation, degradation, human needs, drivers.

Introduction

Soils across the globe are a significant and perhaps the most unheralded category of natural capital. Despite this lack of recognition, it is more important than ever to understand soils’ ecosystem services and natural capital in order to meet the food and fibre demands of a growing global. Many authors (Daily 1997; Swinton *et al.* 2007; Turner and Daily 2008) agree that our ability to understand the ecosystem services and natural capital of our soils is incomplete, despite a good understanding of soil functioning. This paper draws on our understanding of soil science and current ecosystem services thinking to develop a framework for classifying and quantifying soils’ natural capital and ecosystem services. The paper also shows how such framework is being used to build a model of the provision of soils’ ecosystem services on New Zealand dairy farms.

Framework

A new conceptual framework for classifying, quantifying and modelling “soils’ natural capital and ecosystem services” is presented in Figure 1. The framework provides a broad and holistic approach to identify soils’ ecosystem services by: (1) linking soils’ ecosystem services to soils’ natural capital; (2) delineating how soils’ ecosystem services meet “human needs”; (3) identifying how “external drivers” impact on processes that underpin soils’ natural capital. The motivation for seeking to develop this framework lies in the inadequacy of existing frameworks for modelling soils’ services provision. Existing frameworks tend to ignore the scientific knowledge about soils and pay little attention to those factors that managers of soils have control over and therefore have had limited utility as a practical management tool to explore the impacts of land use and practises on the provision of soils’ services. The framework presented here consists of different components:

Natural Capital

Soils are dynamic systems consisting of components (abiotic and biotic) inter-connected by biological, physical and chemical processes that have been well studied by soil scientists. One way of comprehending these complex networks of soil components and their inter-dependencies is to analyse them in terms of the main soil biogeochemical cycles (e.g. water, nitrogen). Another complementary way of characterising and scientifically understanding soils’ natural capital is the concept of soil properties. Properties influence the intensity at which the processes occur and are at the same time products of these processes. In soils, some properties are inherent and cannot be modified and some are more manageable (Lynn *et al.* 2009). Knowing what type of properties is involved in soil functioning, and therefore in the provision of services, is important when it comes to land management and land use changes. For this reason, in putting forward the conceptual framework of soils’ natural capital and ecosystem services (Figure 1), we put a major emphasis on recognising and distinguishing the differences between “inherent” and “manageable” soil properties.

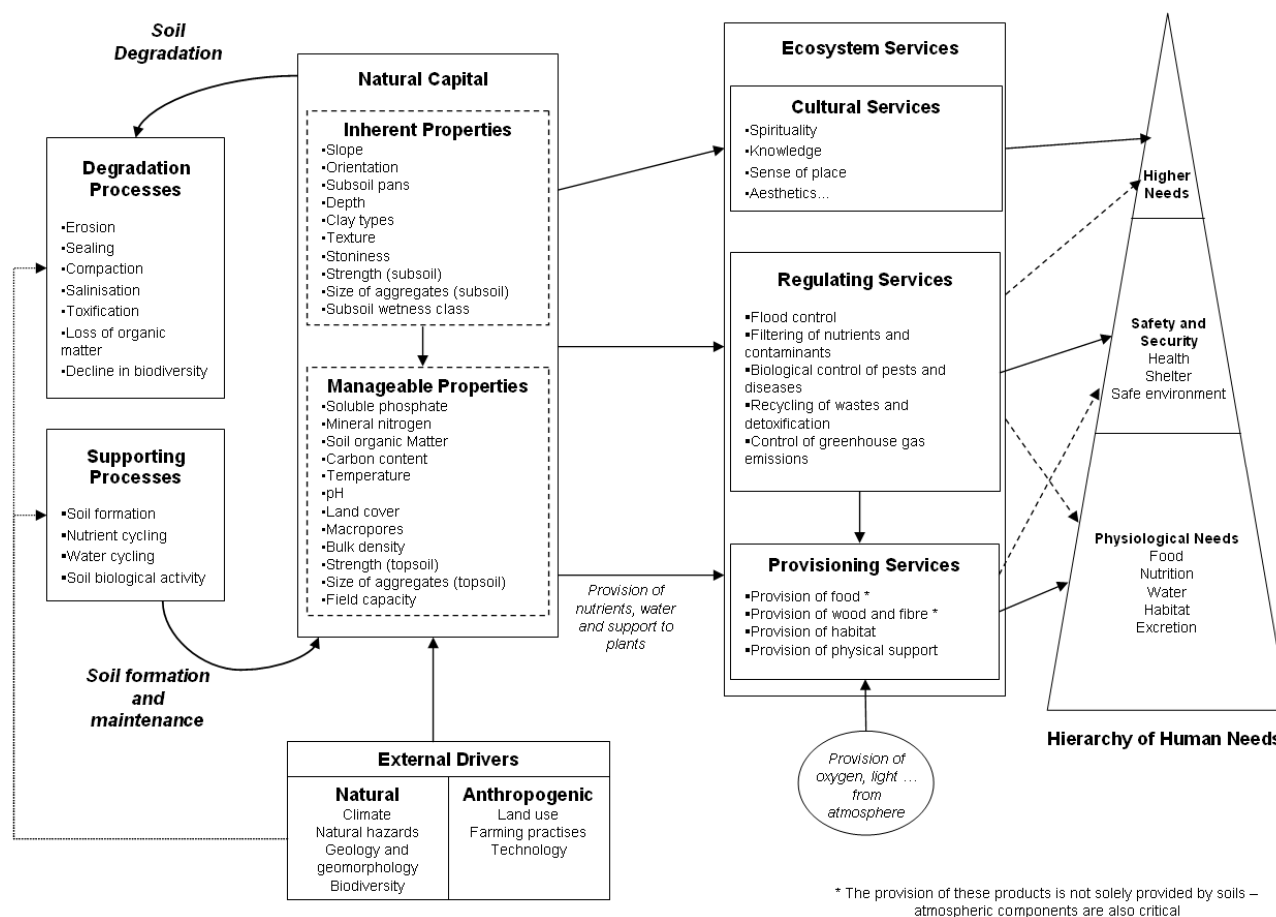


Figure 1. Framework for the provision of Ecosystem services from Soils Natural Capital.

Supporting Processes

We depart slightly from the Millennium Ecosystem Assessment framework (MEA 2005) by distinguishing between “processes” (biological, physical and chemical transformations of input into outputs) and “services” (human well-being derived from natural capital during a given time period). Almost always, an ecosystem service is derived from more than one process. “Supporting soil processes” ensure the formation and maintenance of soils’ natural capital. They are the basic processes that enable soils to exist and function. Supporting processes included in our conceptual framework are: (1) Soil formation: chemical weathering, physical weathering and biological activities, with time, gradually build up and maintain soil properties and ensure the maintenance of the dynamic equilibriums underpinning soils’ natural capital; (2) Nutrient cycling refers to the processes by which a chemical element moves through both the biotic and abiotic compartments of soils. Nutrient cycles maintain equilibriums which drive many processes such as plant uptake, exchange reactions or microbial immobilisation; (3) Water cycling refers to the physical processes enabling water to enter soils, be stored and released. Soil moisture is the driver of many chemical reactions and biological processes and is therefore essential in soil development and functioning; (4) Soil biological activity: Soils provide habitat to millions of species enabling them to function and develop. In return, the activity and diversity of soil biota is essential to soil structure, nutrient cycling and detoxification.

Soil Degradation Processes

Somewhat opposite to “Supporting Processes”, that form and maintain the natural capital of soils, are “Degradation Processes”, which degrade the natural capital of soils (Figure 1). There previously has been very little recognition of ‘degradation’ processes in the soils’ ecosystem services literature (Palm *et al.* 2007). There are many processes that quantitatively and qualitatively degrade soils including: erosion, surface sealing, compaction, salinisation, loss of nutrients, acidification, toxification, loss of organic matter and decline in soil biodiversity. There is a real need to consider these degradation processes in soils’ ecosystem services valuation exercises as the environmental impact they represent may well be greater, or at least, of similar magnitude to soils’ ecosystem services. Furthermore, they are often caused by poor or inappropriate management practices that can in some instances be rectified.

Provisioning Services

According to the MEA (2005, p.40), provisioning services are “products obtained from ecosystems”, including, for example, genetic resources, food and fibre, and fresh water. Soils specifically provide a number of ‘products’ useful for humans, including:

- Provision of habitat for different species: Soils provide complete habitat to different species living in soil such as micro and meso fauna but they also contribute to the provision of a part of the habitat for plants, macro fauna and humans. The provision of habitat by soils is essential because soil biota is the driver of many ecosystem services coming from biological processes.
- Provision of Food, Wood and Fibre: Humans use plants as a source of food, for wood and fibre, as a source of medicines, ornaments, and so forth. For plants to grow they need a number of elements provided by the atmosphere and soils. Soils provide nutrients, water and physical support to plants which strongly affects plant growth.
- Provision of physical support: Soils provide physical support for human infrastructure (e.g. roads), plants and animal species used by humans (e.g. cattle). The integrity of soil structure will affect land use and farming practises.

Regulating Services

Regulating services enable humans to live in a stable, healthy, resilient environment therefore directly contributing to human welfare. Soils’ regulating are therefore:

- Flood control: Soils have the capacity to store and retain quantities of water and therefore can control and lessen the impacts of flood and drought events therefore contributing to the stability and resilience of human habitat.
- Filtering of nutrients and contaminants: Soils have the ability to control water quality by , to some extent, absorbing and retaining solutes and ‘contaminants’, therefore avoiding their release in water bodies such as ground water, lakes and rivers.
- Biological control of pests and diseases: By providing habitat to beneficial species, soils can control the proliferation of pests (crops, animals or humans pests) and harmful disease vectors (e.g. viruses, bacteria).
- Recycling of wastes and detoxification: Soil biota activity degrades and decomposes dead organic matter and destroys chemical compounds that can be harmful to humans.
- Limitation of greenhouse gas emissions: Soils play an important role in regulating the production of greenhouse gases like nitrous oxide (N₂O) and methane (CH₄) therefore impacting on air quality.

Cultural Services

Notably, none of the previous studies (Barrios 2007; Daily 1997; Lavelle *et al.* 2006; Wall *et al.* 2004) on soils’ ecosystem services cover or identify “cultural services”. This is a curious omission as soils alone, as part of landscapes, have across many cultures, always been a source of aesthetic experiences, spiritual enrichment and recreation.

External Drivers

External drivers impact on processes that underpin soils’ natural capital and ecosystem services. External drivers of the soil system are divided into two categories:

- Anthropogenic drivers include: (1) Land use: The type of land use (e.g. cropping, livestock) determines the type of pressures applied to the soil; (2) Farming practises: they determine the level of intensity of the pressures and the amount of inputs to the soil; (3) Technology: new technologies enable humans to manage soil processes to their advantage (e.g. nitrification inhibitors).
- Natural drivers include: (1) Climate: the characteristics of local climate (rainfall intensity, temperature, sunshine) influence supporting processes, degradation processes and biodiversity by driving soil moisture and temperature; (2) Natural Hazards (earthquakes, volcanic eruptions) affect the integrity of soil structure and therefore supporting processes; (3) Geology and geomorphology: the type of parent material determines the original minerals in soils, which will drive soil development and properties. Landscapes are partly determined by geological history; (4) Biodiversity: the type and variety of species present in an area will determine the response of ecosystems, such as soils, to external pressures.

Fulfilling Human Needs

The very essence of the anthropocentric concept of ecosystem services is the fulfilment of human needs. Few studies in the ecosystem services literature, however, go as far as specifying how and what “human needs”

are potentially or actually fulfilled by ecosystem services. One very notable exception is the Millennium Ecosystem Assessment (MEA 2005), that, although not explicitly acknowledging it, shows how ecosystem services contribute to human well-being by using a framework that resembles Maslow's "Hierarchy of needs" (1943). Ecosystem services relate to Maslow's hierarchy of needs on two different levels: (1) The Physical Level: ecosystems provide goods useful for the fulfilment of some physiological need (clean air, clean water, food, shelter), through provisioning processes; (2) The Non-Physical Level: ecosystems provide aesthetics, spiritual and cultural benefits.

Model

From the conceptual framework discussed above we developed a model of the provision of ecosystem services from soils. The model is based on the conceptual framework and uses pedotransfer functions to describe the biophysical processes at the origin of the provision of each services from soils showing how soils Natural Capital is embodied by soil properties and processes. An on-going part of the research is focusing at the farm scale. Scenarios are run on 2 typical New Zealand dairy farms situated on two soils showing very different natural capital: a sedimentary soil and a volcanic soil. The scenarios show how soils' natural capital, farming practises and soil management impact on the provision of ecosystem services from soils. This knowledge can provide management tools for economists and policy makers to better understand the provision of ecosystem services. An economic valuation of soils' services is also undergone. The results of the valuation could be use to build a management tool around payment for ecosystem services, e.g. for farmers, based on soils' natural capital and a better adequacy between land use and available natural capital.

Conclusion

The framework shows how soils' natural capital can be characterised by soil properties, how supporting processes ensure the formation and maintenance of that capital, how degradation processes influence natural capital depletion and how soils' ecosystem services play a role in fulfilling human needs. The framework is then used to develop a biophysical model of the provision of ecosystem services from soils, implemented on New Zealand dairy farms, followed by a valuation of soils' services that could enable economists and policy makers to weigh more carefully soils' natural capital in development processes. It is finally argued that much of the vast scientific modern-day understanding of soil processes and taxonomy needs to be more fully utilised in operationalising frameworks (such as the one proposed in this paper) that attempt to measure soils' ecosystem services and natural capital.

References

- Barrios E (2007) Soil biota, ecosystem services and land productivity. *Ecological Economics* **64**, 269-285.
- Daily GC (1997) 'Nature's Services: Societal Dependence on Natural Ecosystems.' (Island Press: Washington DC).
- Lavelle P, Decaens T, Aubert M, Barot S, Blouin M, Bureau F, Margerie P, Mora P, Rossi JP (2006) Soil invertebrates and ecosystem services. *European Journal of Soil Biology* **42**, S3-S15.
- Lynn I, Manderson A, Page M, Harmsworth G, Eyles G, Douglas G, Mackay A, Newsome P (2009) 'Land Use Capability Survey Hand-book - a New Zealand handbook for the classification of land - 3rd ed.' (Hamilton, Agresearch, Lincoln, Landcare Research, Lower Hutt, GNS Science).
- Maslow AH (1943) A theory of human motivation. *Psychological Review* **50**, 370-396.
- MEA (2005) 'Millennium Ecosystem Assessment: *Ecosystems and Human Well-being: Synthesis*.' (Island Press: Washington DC).
- Palm C, Sanchez P, Ahamed S, Awiti A (2007) Soils: A contemporary perspective. *Annual Review of Environment and Resources* **32**, 99-129.
- Swinton SM, Lupi F, Robertson GP, Hamilton SK (2007) Ecosystem services and agriculture: Cultivating agricultural ecosystems for diverse benefits. *Ecological Economics* **64**, 245-252.
- Turner RK, Daily GC (2008) The ecosystem services framework and natural capital conservation. *Environmental & Resource Economics* **39**, 25-35.
- Wall DH, Bardgett RD, Covich AP, Snelgrove PVR (2004) The need for understanding how biodiversity and ecosystem functioning affect ecosystem services in soils and sediments. In 'Sustaining biodiversity and ecosystem services in soils and sediments'. (Ed. DH Wall) pp. 1-12. (Island Press: Washington, DC).

Mycorrhizal response of halophytes to plant growth in non-saline soil conditions

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Abstract

A glycophyte species and three different halophytes species were examined for colonization and growth response to arbuscular mycorrhizal (AM) fungi under non-saline soil conditions. Roots of *Trifolium alexanderium* as a glycophyte and AM responsive plant highly colonized (more than 85%) and plant growth increased significantly by inoculation with AM fungi. A low level of AM colonization (less than 5%) was found in the roots of *Haloxylon persicum*, *Seidlitzia rosmarinus* and *Salsola sp* as halophytes. Despite low level colonization, AM fungi inoculation increased plant growth of two halophytes species (*Haloxylon persicum* and *Seidlitzia rosmarinus*). The present study demonstrates that although roots of halophytes have intrinsically low symbiosis with AM fungi, but, even at low level of AM colonization, they show positive growth response to AM fungi inoculation in non-saline soil conditions.

Key Words

Arbuscular fungi, halophytes, soil salinity, response.

Introduction

Among the soil microorganisms, arbuscular mycorrhizal (AM) fungi are ubiquitous and are the most widespread mutualistic symbiosis on earth. Mycorrhizal fungi form a mutually beneficial symbiosis with most terrestrial plants and occur in the soil of most ecosystems (Smith and Read 2008). They play an important role in nutrient cycling and benefit plants in terms of their growth and soil structure development. Arbuscular mycorrhizal fungi occur in many stressful environments. Although relatively large populations of AM fungi spores have been found in saline conditions (Aliasgharzadeh *et al.* 2001; Sengupta and Chaudhuri 1990; Wang *et al.* 2004), increased salinity decreases AM spore population, root colonization and hyphal extension (Juniper and Abbott 2006; Peat and Fitter 1993). Different levels of AM colonization in halophytes have been reported in many field studies in different locations with different soil salinity levels (Asghari *et al.* 2008). Salinity could reduce AM colonization by directly reducing hyphal growth and/or decreasing plant growth (less carbohydrate). But the most important reason of low levels of AM colonization in halophytes is the detrimental effect of salinity on hyphal growth. A recent report indicates that the most important effect of salinity on AM fungi is related to its detrimental effect on spore germination and hyphal production (Juniper and Abbott 2006). It is not clear if reduced soil salinity may develop mycorrhizal benefits in halophytes. The objective of this study was to evaluate AM colonization of halophytes and mycorrhizal response of halophytes to plant growth under non-saline conditions.

Material and methods

An experiment was carried out to investigate the effects of AM fungi on plant growth under non saline soil conditions in four different plant species. Plants were *Trifolium alexanderium* as a mycorrhizal responsive species, *Haloxylon persicum*, *Seidlitzia rosmarinus* and *Salsola sp* as halophytes and non mycorrhizal responsive species. Plants were grown in a soil with low level of salinity (0.4 dS/m) under green house conditions for 12 weeks. The soil was autoclaved (110 °C, 1 h, twice at 48 h intervals) to remove indigenous AM fungal propagules. Pots (2 kg) were inoculated with *Glomus intraradices* Schenk and Smith (DAOM 181602) or not inoculated. Pots containing sterilized soil received a filtrate from 5 g of original non-sterile soil, to reintroduce a soil microflora (without AM fungi). Long Ashton nutrient solution without P was added (10 mL per pot) to the pots once per week for 8 weeks. The experiment had a randomized complete block design with 2 treatments (AM inoculated and non AM inoculated). There were six replicates per treatment. Data were analyzed by T-Test.

Results

Roots of *Trifolium alexanderium* plants were highly colonized by AM fungi (more than %85), but less than %5 colonization were found in halophytes. Mycorrhizal inoculation significantly increased shoot dry weight of *Trifolium alexanderium*. Despite low level of AM colonization, *Haloxylon persicum* and *Seidlitzia rosmarinus* shoot dry weights were increased by AM fungi inoculation (Figure 1)

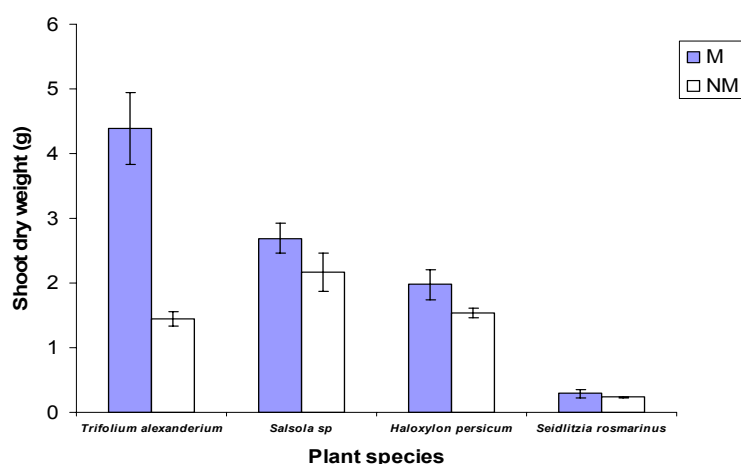


Figure 1. Shoot dry weight of AM fungi inoculated (M) and non-inoculated (NM) *Trifolium alexanderium*, *Salsola sp*, *Haloxylon persicum* and *Seidlitzia rosmarinus* after 12 weeks. Vertical bars represent standard error of the means, n =6.

Conclusion

The detrimental effects of soil salinity on spore germination and hyphal growth of AM fungi have been identified in past studies as being the most important reason for the absence of AM fungi colonization in halophytes. In contrast to this, we found that reducing soil salinity did not improve AM colonization in halophytes. Thus, it is likely that other factors such as environmental factors, phenology of host plant (Wilson and Hartnett 1998) and other soil properties (Caroline and Bagyaraj 1995; Mamatha *et al.* 2002) may affect colonization by AM fungi in halophytes. Despite of low level of AM colonization, halophytes show a positive growth response to AM inoculation in non-saline soil conditions. More work to determine factors that promote AM colonization in halophytes is required.

References

- Aliasgharzadeh N, Rastin NS, Towfighi H, Alizadeh A (2001) Occurrence of arbuscular mycorrhizal fungi in saline soils of the Tabriz Plain of Iran in relation to some physical and chemical properties of soil. *Mycorrhiza* **11**, 19–122.
- Asghari HR, Amerian M, Gorbani H (2008) Soil salinity affects arbuscular mycorrhizal colonization of halophytes. *Pakistan Journal of Biological Science* **11**, 1909–1915.
- Caroline M, Bagyaraj D J (1995) Mycorrhization helper bacteria and their influence on growth of cowpea. In ‘Mycorrhiza.: Biofertilizers for the Future’. (Eds A Adholeya, S Singh) pp. 192–196. (TERI: New Delhi).
- Juniper S, Abbott LK (2006) Soil salinity delays germination and limits growth of hyphae from propagules of arbuscular mycorrhizal fungi. *Mycorrhiza* **16**, 371–379.
- Mamatha G, Bagyaraj DJ, Jaganath S (2002) Inoculation of field-established mulberry and papaya with arbuscular mycorrhizal fungi and a mycorrhiza helper bacterium. *Mycorrhiza* **12**, 313–316.
- Peat HJ, Fitter AH (1993) The distribution of arbuscular mycorrhizas in British flora. *New Phytologist* **125**, 845–854.
- Sengupta A, Chaudhuri S (1990) Vesicular arbuscular mycorrhiza (VAM) in pioneer salt marsh plants of the Ganges river delta in West Bengal (India). *Plant and Soil* **122**, 111–113.
- Smith SE, Read DJ (2008) ‘Mycorrhizal symbiosis’, 3rd edition. (Academic Press: London, UK).
- Wang FY, Liu RJ, Lin XG, Zhou JM (2004) Arbuscular mycorrhizal status of wild plants in saline-alkaline soils of the Yellow River Delta. *Mycorrhiza* **14**, 133–137.
- Wilson GWT, Hartnett DC (1998) Interspecific variation in plant responses to mycorrhizal colonization in tallgrass prairie. *American Journal of Botany* **85**, 1732–1738.

Nitrogen leaching in soil amended with biochars produced at low and high temperatures from various feedstocks

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Abstract

Leaching losses of nitrogen (N) from agricultural land are a major source of water pollution in the United States. A greenhouse trial of biochars produced from corn stover, oak wood, paper mill waste, and poultry manure mixed with sawdust at 300°C and 600°C was conducted in order to evaluate the potential of these biochars to reduce N losses to the environment. Nitrate (NO₃) leaching was reduced in the 7% biochar treatment for both 300°C oak and 300°C paper mill waste, while NO₃ leaching increased in the 7% biochar treatment for 300°C poultry manure with sawdust. Ammonium (NH₄) leaching was not reduced in any of the 300°C biochar treatments and an increase was observed in the 300°C corn, paper mill waste, and poultry manure with sawdust treatments. NO₃ leaching was reduced with increasing application rates of 600°C corn and oak biochars. NH₄ leaching was slightly reduced in the 2% and 7% biochar treatments for 600°C paper mill waste and 600°C oak and increased in the 7% biochar treatment for 600°C poultry manure and sawdust treatment. The effect of higher biochar application rates, successive cropping years, and multiple soil types should be investigated in future studies.

Keywords: biochar, nitrate leaching, ammonium leaching, biochar feedstock, nitrate, ammonium.

Introduction

N loss from agricultural soils by leaching can lead to both declines in crop yields and negative environmental impacts including eutrophication of surface waters (Brady and Weil 2008). In addition to the management of vegetation and fertilization, biochar application to soil has been shown to affect the leaching of N through multiple mechanisms (Major *et al.* 2009). In addition to the preexisting physical, chemical, and biological characteristics of the soil to which biochar is applied, the feedstock from which biochar is produced, as well as the temperature at which biochar is produced are important factors related to the effectiveness of a given biochar. Although biochar has been shown to increase soil fertility and carbon sequestration (Glaser *et al.* 2002; Lehmann *et al.* 2003; Lehmann *et al.* 2006), insufficient knowledge exists related to the suitability of pyrolysis feedstocks for biochar production, as well as the ideal pyrolysis conditions for a given pyrolysis feedstock for biochar production (Chan *et al.* 2008).

Methods

Biochar feedstock selection and pyrolysis

An assessment of regionally abundant and available feedstocks was conducted in order to identify the most economically promising pyrolysis feedstocks from which to produce biochar in New York State, USA. These feedstocks included corn stover, oak wood, paper mill waste, and poultry manure mixed with sawdust. The feedstocks were collected, dried to below 10% moisture content, and pyrolyzed using a slow pyrolysis method at a range of temperatures.

Soil type and preparation

A greenhouse trial was conducted in order to evaluate the potential NO₃ and NH₄ leaching reductions achievable with biochars produced from distinct feedstocks at low and high temperatures. True soil was used in the greenhouse trial in order to mimic the biochar-soil interactions that occur in the field. Soil used in the greenhouse trial was Junius loam, Kendaia silt loam, and Lima loam, that was collected from a university research farm in Aurora, NY, USA. The soil pH was 6.85. The soil was continuously cropped in corn for over 50 years preceding collection. The soil was transported to the greenhouse, where it was air-dried. The dried soil was then shredded to break apart clods and sieved to 2mm to maintain uniform particle size. In order to determine the volume of water that was needed to exceed the soil water holding capacity and initiate leaching, the soil water holding capacity was determined. Biochars produced at 300°C and 600°C from corn stover, oak wood, paper mill waste, and poultry manure mixed with sawdust were mixed with field soil at application rates of 0.2, 0.5, 2, and 7% (w/w) using a V mixer and poured into pots. There were 2 replicates of each treatment.

Crop variety and greenhouse conditions

All pots were randomly spaced throughout the greenhouse bench system. Rootworm resistant hybrid corn treated with fungicide was planted. Starter fertilizer was applied at a rate of 2.02 kg N / ha and later supplemented with 22.03 kg N / ha in order to meet crop fertilizer recommendations for the soil used in this trial. Watering was performed when leaf curling was observed in order to avoid hypoxia. The greenhouse temperature was maintained at 24°C during the day and 18°C at night.

Leaching experiment and leachate analysis

All leachate generated during the trial flowed into an enclosed collection system and was collected approximately 24 hours after each leaching event. The leachate was taken from the greenhouse to the laboratory where its volume was recorded and a cumulative subsample (5% by volume) was removed and frozen. After completion of the trial, subsamples were analyzed for NO₃ and NH₄ using colorimetry on a segmented flow analyzer.

Results

NO₃ and NH₄OH leaching in the control pots was observed to be 31.78 kg NO₃/ha and 0.21 kg NH₄OH/ha, respectively. NO₃ leaching was reduced in the 7% biochar treatment to 16.40 and 11.89 kg NO₃/ha for 300°C oak biochar and 300°C paper mill waste biochar, respectively, while NO₃ leaching increased in the 7% biochar treatment for 300°C poultry manure with sawdust biochar to 63.71 kg NO₃/ha (Fig. 1). NH₄ leaching was not reduced in any of the 300°C biochar treatments and an increase in the 7% biochar treatments to 1.43 kg NH₄OH/ha was observed in 300°C corn biochar, 1.08 kg NH₄OH/ha in 300°C paper mill waste biochar, and 1.28 kg NH₄OH/ha in 300°C poultry manure with sawdust biochar treatments (Fig. 2). NO₃ leaching was reduced with increasing application rates of 600°C corn biochar to 6.46 kg NO₃/ha and oak biochar to 15.99 kg NO₃/ha (Fig. 3). NH₄ leaching was slightly reduced in the 2% and 7% biochar treatments for 600°C paper mill waste biochar to 0.04 kg NH₄OH/ha and 0.02 kg NH₄OH/ha, respectively (Fig. 4) NH₄ leaching was reduced in the 0.5%, 2%, and 7% biochar treatments for 600°C oak biochar to 0.03, 0.07, and 0.06 kg NH₄OH/ha, respectively. NH₄ leaching increased in the 7% biochar treatment for 600°C poultry manure and sawdust biochar to 0.62 kg NH₄OH/ha (Fig. 4).

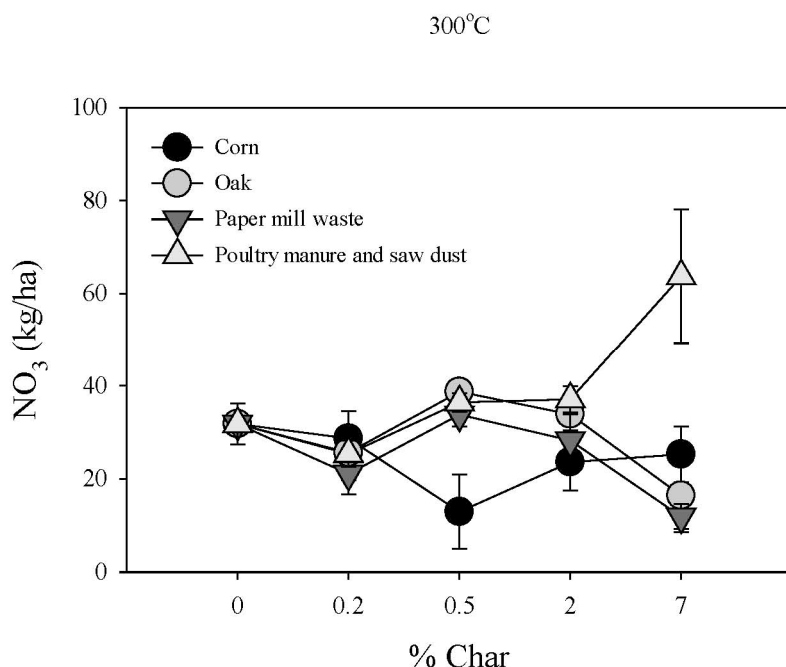


Figure 1. NO₃ lost as leachate from soil amended with biochar produced from various feedstocks at 300°C.

300°C

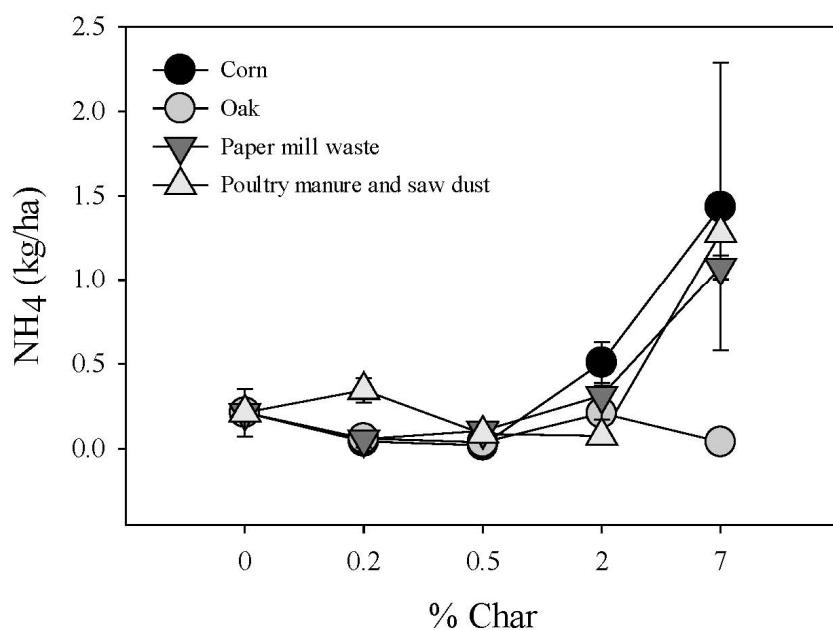


Figure 2. NH_4 lost as leachate from soil amended with biochar produced from various feedstocks at 300°C.

600°C

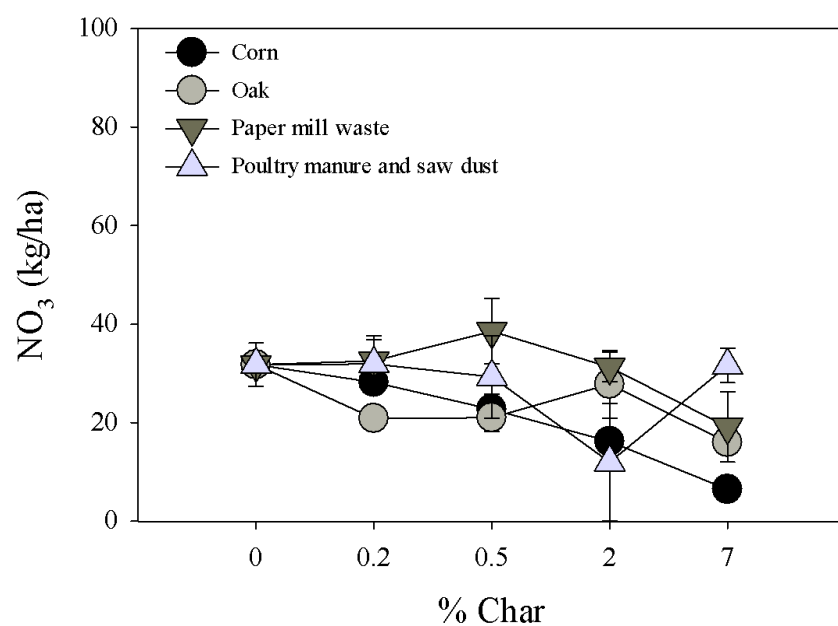


Figure 3. NO_3 lost as leachate from soil amended with biochar produced from various feedstocks at 600°C.

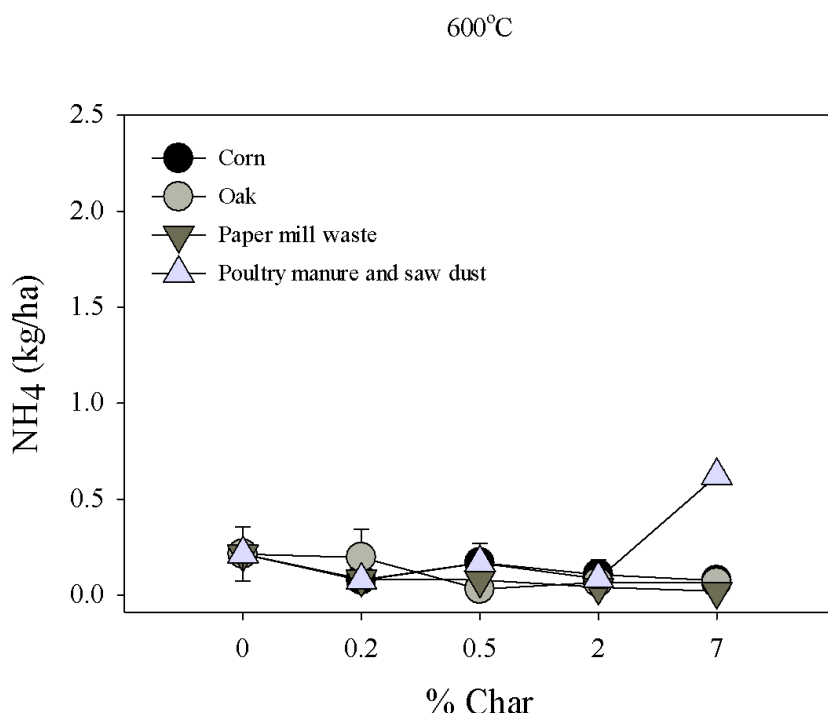


Figure 4. NH₄ lost as leachate from soil amended with biochar produced from various feedstocks at 600°C.

Conclusion

The results of this study showed that biochar application rates, biochar feedstock, and pyrolysis temperature affect the loss of NO₃ and NH₄ from soil by leaching. In order to effectively minimize N losses to the environment, it is necessary to evaluate each biochar individually at multiple application rates.

References

- Brady NC, Weil RR (2008) 'The Nature and Properties of Soils.' (Prentice Hall: Upper Saddle River, NJ)
- Chan KY, Van Zwieten L, Meszaros I, Downie A, Joseph S (2008) Using poultry litter biochars as soil amendments. *Australian Journal of Soil Research* **46**, 437-444.
- Glaser B, Lehmann J, Zech W (2002) Ameliorating physical and chemical properties of highly weathered soils in the tropics with charcoal – a review. *Biology and Fertility of Soils* **35**, 219-230.
- Lehmann J, de Silva JP Jr, Steiner C, Nehls T, Zech W, Glaser B (2003) Nutrient availability and leaching in an archaeological Anthrosol and a Ferralsol of the Central Amazon Basin: fertilizer, manure and charcoal amendments. *Plant and Soil* **249**, 343-357.
- Lehmann J, Gaunt J, Rondon M (2006) Bio-char sequestration in terrestrial ecosystems – a review. *Mitigation and Adaptation Strategies for Global Change* **11**, 403-427.
- Major J, Steiner C, Downie A, Lehmann J (2009) Biochar effects on nutrient leaching. In 'Biochar for environmental management'. (Eds. J Lehmann, S Joseph) pp. 271-287. (Earthscan: London)

Non-target impacts of the nitrification inhibitor dicyandiamide on soil biota

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Abstract

Intensively grazed dairy pastures in New Zealand routinely receive concentrated inputs of nitrogen (N) both through application of fertiliser and deposition of urine by grazing animals. Leaching of nitrate from soil into aquifers, rivers and lakes is a source of increasing environmental and public health concern. Nitrification inhibitors have been shown to decrease leaching and denitrification from urea- and ammonium-based fertilisers and from urine patches in pastures. To date there have been few studies on effects of nitrification inhibitors on non-target soil biota. Three laboratory experiments were carried out to measure the short term effects of a nitrification inhibitor, dicyandiamide (DCD), on diversity of soil bacterial populations, earthworms and Collembola. Molecular analysis of the soil bacterial community indicated that application of DCD to soil did not affect the composition of the predominant bacterial phyla present in soil, unlike the addition of bovine urine which caused rapid changes in bacterial diversity. Survival and growth of the earthworm *Aporrectodea caliginosa* was unaffected by application of urine or DCD to soil. Collembola populations were not inhibited by DCD, while the addition of urine appeared to increase numbers. Results confirm the view that DCD can be considered one of the more environmentally benign nitrification inhibitors, making it an important tool in countering the environmental impacts resulting from ongoing land use intensification.

Key Words

Environmental impact, mitigation, SARST, soil quality, soil bacterial communities.

Introduction

Intensification of New Zealand's farming systems has resulted in an increase in fertiliser inputs, nutrient recycling rates and stocking densities. Intensively grazed dairy pastures in New Zealand routinely receive large inputs of nitrogen (N) both through application of fertiliser and deposition of urine by grazing animals. Several studies have indicated that, at least in New Zealand, animal urine patches are the major source of N leached from grazed pastures (Ledgard *et al.* 1999). Leaching of nitrate from soil into aquifers, rivers and lakes is causing severe environmental impacts (apparent in eutrophication of some of New Zealand's iconic lakes, such as Lake Taupo) and public health concerns.

Nitrification inhibitors are being used increasingly in New Zealand to reduce N loss from soil and increase N use efficiency. The inhibitors have been shown to decrease leaching from urea- and ammonium-based fertilisers and from urine patches in grazed pastures (e.g. Di *et al.* 2007). However, there have been few independent studies on the non-target effects of these now widely used agricultural compounds. Cuttle (2008) concluded that there appeared to be no evidence of wider environmental impacts arising from the use of nitrification inhibitors but after reviewing the available literature, Edmedes (2004) stated that research was needed to quantify the effects (both long and short term) of these chemicals, and their repeated use, on soil quality. Given the importance of maintaining soil quality and function for ongoing productivity of pastoral systems, it is essential that soil treatments such as nitrification inhibitors do not impact on activity of non-target soil microbes and fauna.

This contribution reports on results of studies carried out to determine the short term effects of the nitrification inhibitor dicyandiamide (DCD, C₂H₄N₄), which is used in New Zealand, on soil bacterial diversity and representative soil biota. The Collembolan species *Folsomia candida* has frequently been used as an indicator species in ecotoxicological testing and is highly sensitive to intensive use of pesticides. The earthworm species chosen for experiments, *Aporrectodea caliginosa*, is commonly found in New Zealand pastures and is likely to be exposed to DCD through standard agricultural practice.

Methods

Pot trials

Three laboratory experiments were carried out to examine impacts of DCD on 1) soil bacterial community; 2) earthworms; 3) Collembola. For assessment of impacts of DCD on the soil bacterial community, a pot trial was established. Pots containing 1 kg of field moist soil (Horotiu sandy loam) were sown with ryegrass seedlings (*Lolium perenne*) and maintained in a controlled-environment (CE) room (12°C day/8°C night) following application of treatments. For assessment of impacts of DCD on Collembola and earthworms, planter bags containing 500g of field moist soil (Wakanui loam) were sown with ryegrass. The planter bags were maintained in a shade house and the ryegrass was allowed to grow for six weeks before treatments were applied to the soil surface and soil was removed to containers in a CE room, as detailed below.

Treatments

Cow urine was collected from a dairy farm near Hamilton, New Zealand. Pots/planter bags were treated with equivalent of 600 kg/ha of urine-N (within the range of N concentration typically found under urine patches) and equivalent of 30 kg of DCD/ha, a rate typical of the amount applied to pasture in New Zealand. Treatments applied were: urine; DCD, urine+DCD and an untreated control.

Molecular analysis of soil bacterial community

Microbial community nucleic acids were extracted from soil samples collected 2 and 56 days after application of treatments as described previously (Griffiths *et al.* 2000). Genetic diversity of the soil bacterial community was profiled by serial analysis of ribosomal sequence tags (SARST) as described by Yu *et al.* (2006). The taxonomic identity of aligned 16S sequences from each treatment was determined using the classify tool within greengenes (DeSantis *et al.* 2006). Because of the relatively short sequences obtained using SARST, classifications were made to the level of phylum only.

Earthworms

After treatments were applied, the ryegrass was cut to approximately 1cm and the trimmings discarded. The contents of the bags were then mixed to incorporate the remaining grass and roots into the soil, with a subsample of 500g from each planter bag transferred into lidded plastic containers. *A. caliginosa* were collected from the field and held in field soil at 15°C before use. Earthworms were individually weighed and then grouped in sets of three, so that the average weight of the three worms was similar for all groupings. Three earthworms were added to each container and the containers were maintained in a CE room at 12°C. Earthworm survival, development (weight) and sexual maturity (presence of a clitellum) was assessed every 2 weeks for 10 weeks. After each assessment the worms were returned to the soil, the soil moistened as necessary and food (dried, powdered cow dung and grass) was added. Weight data was analysed in Genstat (ANOVA), while sexual maturity was analysed using a generalised linear mixed model.

Collembola

As for the earthworms, the treated soil in the bags was mixed and subsamples of 150g were transferred to a lidded plastic containers with two replicates prepared from each planter bag. Ten adult Collembola from a laboratory culture were added to each container and containers were maintained in an incubator at 15°C. Collembola populations were assessed after 28 and 56 days by destructively sampling two replicate containers per treatment, and analysis of variance was performed to compare standard error of the means for each assessment time.

Results

Soil bacterial diversity

Analysis of sequence data showed that bacterial diversity in soil treated with DCD did not differ significantly from untreated control soil at either 2 or 56 days after soil treatment (Figure 1). The proportions of the predominant phyla present in untreated soil, Proteobacteria, Acidobacteria, Actinobacteria and Firmicutes, were unchanged following application of DCD to soil. In contrast, there was rapid shift in the microbial community in response to urine and urine+DCD treatment. Shifts in community structure 2 days after application of treatments could mainly be attributed to increased members of the phylum Firmicutes, which comprised approximately 10% of the total phyla in control and DCD treated soils, in comparison with approximately 50% in the urine and urine+DCD treated soils.

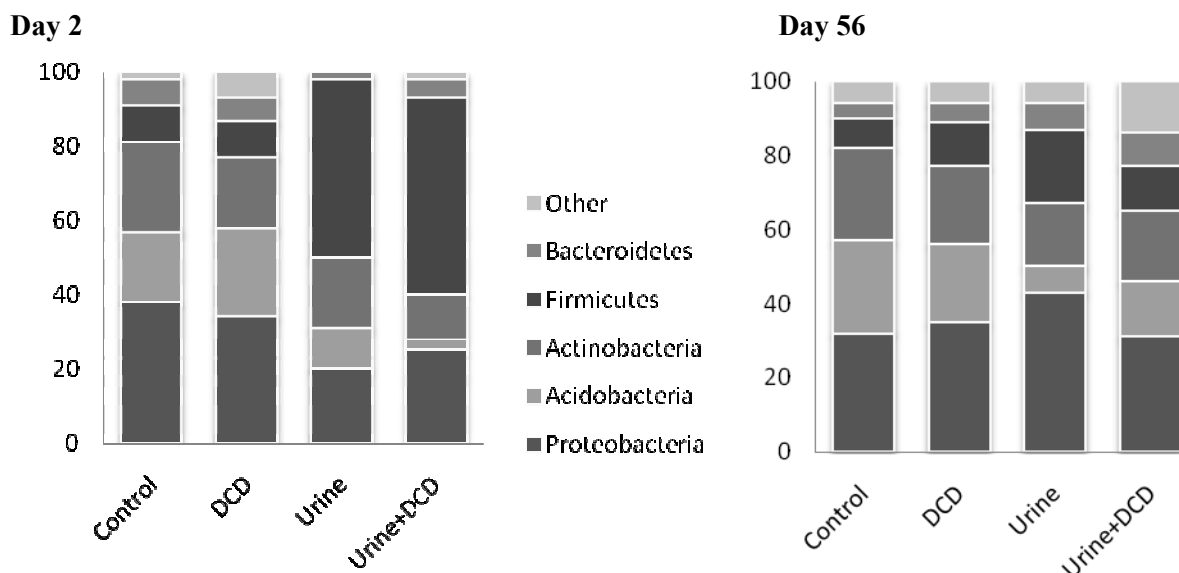


Figure 1. Dominant taxa at a phyla level based on SARST analysis of 16S rRNA sequences, in untreated soil and in soils treated with urine, DCD, and urine+ DCD, 2 and 56 days after the application of treatments (based on an average of 560 OTUs from each sample).

Earthworms

Survival and development of the earthworms was unaffected by any of the treatments. Earthworms increased in weight during the 10 weeks of the experiment (Figure 2), apart from a slight decrease at week four across all treatments, when they received a smaller amount of food. Treatment had no effect on the maturation rate of the earthworms (data not shown).

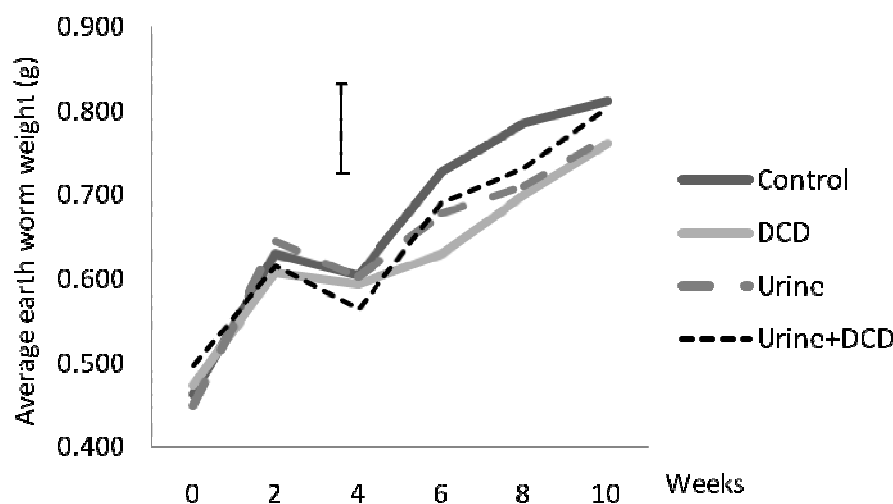


Figure 2. Weight of earthworms incubated in soil treated with urine, DCD and urine+ DCD. Error bar is SEM.

Collembola

Presence of DCD in soil had no significant effect on populations of *F. candida*, with populations similar to those in untreated soil at 28 and 56 days after treatment. Addition of urine to the soil appeared to have a positive effect on Collembola populations, with larger populations recovered from urine and urine+DCD treatments than the control after 28 days, although this difference was only significant ($P < 0.05$) for the urine+DCD treatment. There were no significant differences between treatments after 56 days.

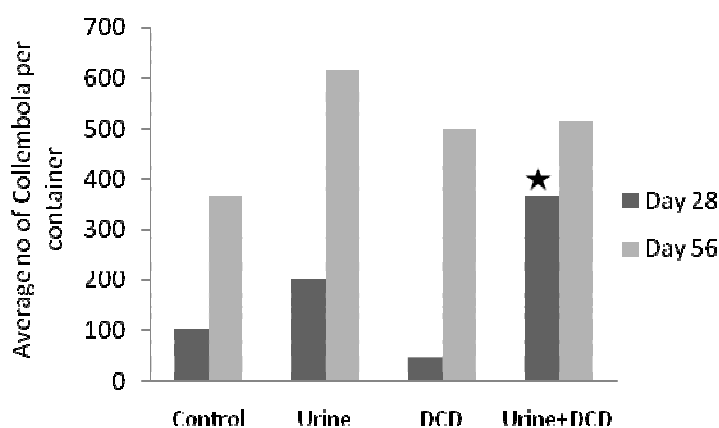


Figure 3. Numbers of the Collembolan *Folsomia candida* in soils at 28 and 56 days after application of urine, DCD, and urine+ DCD. Star indicates significant difference between Control and Urine+DCD treatments at Day 28.

Discussion

DCD is considered one of the more environmentally benign nitrification inhibitors. Certainly SARST analysis of bacteria diversity at days 2 and 56 indicated that there was no short term effect on the predominant phyla present in soil. The current analysis was limited to the level of phylum, and it is possible that changes in the bacterial community were occurring beneath this level. However, given that the inhibitor is targeted at activity of a specific functional group of bacteria, the ammonia-oxidisers, these results add further weight to the conclusion that DCD has little impact on non-target microflora. Similarly, DCD had no impact on earthworm growth rate and survival and fecundity of a Collembola population. Both Collembola and earthworms are regarded as key “ecosystem service providers”, as they play important roles in maintaining the soil ecosystem, with Collembola in particular being regarded as key indicators of soil fertility and health. The use of chemical inhibitors to disrupt natural soil processes seems at first glance to be contrary to ecological concepts of soil quality and conservation of biodiversity. However, given its low level of non-target impact on soil microflora and fauna, application of DCD to pasture is a relatively benign intervention that has an important role to play in mitigating the environmental hazards imposed by ongoing land use intensification.

References

- Cuttle SP (2008) Impacts of pastoral grazing on soil quality. In ‘Environmental Impacts of Pasture-Based Farming’. (ed RW McDowell), pp. 93-108. (CABI: Wallingford).
- DeSantis TZ, Hugenholtz P, Larsen N, Rojas M, Brodie EL, Keller K, Huber T, Dalevi D, Hu P, Andersen GL (2006) Greengenes, a chimera-checked 16S rRNA gene database and workbench compatible with ARB. *Applied and Environmental Microbiology* **72**, 5069-5072.
- Di HJ, Cameron KC, Sherlock RR (2007) Comparison of the effectiveness of a nitrification inhibitor, dicyandiamide, in reducing nitrous oxide emissions in four different soils under different climatic and management conditions. *Soil Use and Management* **23**, 1-9
- Edmeades DF (2004) Nitrification and urease inhibitors: A review of the national and international literature on their effects on nitrate leaching, greenhouse gas emissions and ammonia volatilisation from temperate legume-based systems. Environment Waikato Technical Report 2004/22. ISSN: 1172-4005 17 pp.
- Griffiths RI, Whiteley AS, O'Donnell AG, Bailey MJ (2000) Rapid method for coextraction of DNA and RNA from natural environments for analysis of ribosomal DNA and rRNA-based microbial community composition. *Applied and Environmental Microbiology* **66**, 5488-5491.
- Ledgard SF, Penno JW, Sprosen MS (1999) Nitrogen inputs and losses from clover/grass pastures grazed by dairy cows, as affected by nitrogen fertilizer application. *Journal of Agricultural Science* **132**, 215-225.
- Yu Z, Yu M, Morrison M (2006) Improved serial analysis of V1 ribosomal sequence tags (SARST-V1) provides a rapid, comprehensive, sequence-based characterization of bacterial diversity and community composition. *Environmental Microbiology* **8**, 603-611.

Reactive nitrogen cycling and potential ecosystem services trade-offs in an eastern Corn Belt soil

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Abstract

Outcome of ecosystem services assessments across a range of agroecosystems can be governed by management choices. Additionally, systematic nitrogen (N) additions in agricultural fields can pose a crucial dilemma as N inputs are needed to achieve yield goals, but they also typically cause increases in edge-of-field N losses that frequently deteriorate both water and air qualities. This study examines potential ecosystem services trade offs in typical US Corn Belt managed ecosystems. We quantify long-term nitrate (NO_3^-) loads in subsurface drains, and nitrous oxide (N_2O) fluxes at soil surface in corn (*Zea mays* L.) monocultures (CC) receiving annually-repeated N inputs of spring- or fall-applied liquid swine manure (SM and FM, respectively), or spring-applied urea-ammonium nitrate (UAN), and in an unfertilized, restored prairie grass (PG). Losses of both N_2O and NO_3^- from PG were negligible. The greatest N_2O emissions occurred in CC's receiving either UAN or SM (6.4-8.2 kg N_2O -N /ha/yr), while CCFM exhibited an intermediate magnitude (3.3 kg N_2O -N /ha/yr). Conversely, CCFM increased NO_3^- losses by roughly 1.7-fold relative to all other CC treatments (33 vs. 19 kg NO_3^- -N /ha/yr) revealing a clear directional trade off between N_2O and NO_3^- outcomes when shifting between FM and SM managements.

Key Words

Ecosystem resilience, soil quality, life cycle assessment, provisioning, regulating, and supporting services.

Introduction

Conceptually, “ecosystem services” refers to any benefit that human society can derive from natural or managed ecosystems (Tilman 2002; Daily and Matson 2008). Ecosystem services are typically classified as provisioning (e.g., water, grain, biomass, energy), supporting (e.g., nutrient cycling and dispersion), regulating (e.g., climate regulation, water purification), and cultural types (e.g., recreational, ecotourism). Within conventional agricultural systems, management choices have been disproportionally made toward enhancing provisioning services (i.e., harvested grain), and performance assessments for these systems have been typically based only on achievement of pre-established yield goals (Tilman 2002). Consequently, other potentially beneficial ecosystem functions such as efficient nutrient cycling and removal of contaminants from the environment are usually ignored and/or neglected resulting in detrimental outcomes for these ecosystem services (Palmer and Filoso 2009; Vitousek 2009). The dilemma regarding these apparently competing ecosystem services becomes even more critical in the case of N cycling as N inputs are typically key factor for sustaining high productivity in agroecosystems, but the associated increases in N losses to the atmosphere (e.g., N_2O) and surface and groundwater bodies (e.g., NO_3^-) can substantially diminish environmental quality. Lately, it is being recognized the need to use methodologies that would allow to comprehensively assess all the known ecosystem services and environmental consequences in a given scenario. This information would be useful for decision making processes based on the premise that management choices can alter the overall outcome of ecosystem services from agroecosystems. Basic steps in these balanced, integral assessments include to enhance data availability as well as to assess the directional outcomes of multiple ecosystem services. Thus, the objective of this study was to examine the potential ecosystems services trade offs within typical US Corn Belt managed ecosystems particularly focusing on NO_3^- vs. N_2O losses from soils as well as other associated components of the terrestrial reactive N cycling such as soil and plant N pools and grain yield.

Materials and methods

This study was conducted at the Water Quality Field Station located at Purdue University ACRE farm, West Lafayette, IN (40°29'55" N, 86°59'53" W, 215 m elevation). Soil series are Drummer silty clay loam (fine-silty, mixed, superactive, mesic Typic Endoaquoll) and Raub silt loam (fine-silty, mixed, superactive, mesic Aquic Argiudoll). Between 1998 and 2006, we quantified NO_3^- effluxes at subsurface drains by determining both drainage water flow and dissolved NO_3^- concentrations using conventional colorimetric analytical methods, corn grain yield, N pools in both corn plant and surface soil (Hernandez-Ramirez *et al.* 2009b), and

soil surface N₂O emissions as point measurements using static chamber (Hernandez-Ramirez *et al.* 2009b). The experimental site has 12 treatments in a randomized complete block design with four replicates and experimental plots with dimensions of 10.8 × 48 m². Treatment arrangement includes four factors: managed ecosystem, N source, rate, and timing. The managed ecosystems were corn-soybean [*Glycine max* (L.) Merr.] rotation with sets of experimental plots for both crops (corn: CS and soybean: SC) in any given year, continuous corn (CC), and an unfertilized restored prairie grass (PG) dominated by big bluestem (*Andropogon gerardii* Vitman) served as a control. The N sources for corn treatments were UAN [28% (w/w) N] injected either in preplant (PP) or side-dressed (SD) at corn growth stage V5, and liquid swine manure [C/N ratio: 2:1, 80% (w/w) of N as NH₄⁺] injected into CC at a rate of 255 ± 24 kg N /ha/yr in either the spring (SM) or the fall (FM). The UAN application rates and timings for CS treatments were 135 at SD, and 157 at PP and 180 at PP, and for CC treatments were 157 at SD, and 180 at PP and 202 kg N /ha/yr at PP. Both UAN and manure were placed at a depth of 0.10 m in the soil. The resulting treatment combinations were PG, CCSM, CCFM, CCUAN 157SD, CCUAN 180PP, CCUAN 202PP, CSUAN 135SD, SC (~ 135SD), CSUAN 157PP, SC (~ 157PP), CSUAN 180PP, and SC (~ 180PP). To clearly differentiate our three SC treatments, we included in parentheses in their acronyms the N rates and timings of the corresponding three CS treatments in crop rotations in our study. Analyses of variance models and Tukey test at a critical value of 0.05 were run to examine treatment effects.

Results and discussion

Effects of management choices on soil-water-air quality aspects

Environmental quality assessment of the managed ecosystems at this long-term experimental site primarily encompasses quantification of critical components and fluxes within the reactive compartment of the N cycle including NO₃⁻ losses, N₂O emissions as well as soil and plant N pools. Regarding NO₃⁻ leaching to drainage water, one of the most novel, striking findings was the substantially increased NO₃⁻ loss caused by annually-repeated FM additions which resulted in roughly 1.7-fold greater NO₃⁻ losses than SM as well as CCUAN's treatments (33 vs. 19 kg N /ha/yr; *P* < 0.001; Figure 1A). These highest NO₃⁻ losses from CCFM can be directly attributed to abundant precipitation (i.e., rain and snow; data not shown) and the associated major drainage events taking place during the winter and early spring after manure-N additions in the late fall. Another interesting-unique feature of this dataset is that PG restoration system consistently minimized NO₃⁻ quantities resulting in relatively negligible NO₃⁻ losses (2.5 kg N /ha/yr; Figure 1A). This outcome can be explained because PG management consists in N additions, no tillage operations, and annual burning of aboveground biomass residues that could increase gaseous N losses (Hernandez-Ramirez *et al.* 2009a). Tillage and/or N additions would have stimulated organic matter mineralization and associated NO₃⁻ production in these soils. In addition, PG typically has a longer growing season than maize cultivation resulting in greater overall water uptake and a consequent decreased percolation in PG fields as well as increased N uptake and storage by PG perennial roots (Huggins *et al.* 2001). Hence, these several combined factors likely suppressed NO₃⁻ losses from PG fields. It is also remarkable the changes in magnitude of NO₃⁻ losses between the corn (CS) and soybean (SC) phases of these crop rotations (Figure 1B). Irrespective of UAN rate and time of N addition, all three corn-soybean rotations registered approximately 15% increased NO₃⁻ losses in SC relative to CS (*P* < 0.001). This pattern can be explained by carry over effects from a given corn growing season into the immediately following fall-to-spring period before soybean cultivation. High soil residual NO₃⁻ after corn years (Kaspar *et al.* 2007) and lower soil N uptake by soybean can be concomitant causes for enhanced NO₃⁻ exportation from these fields within SC phase. As expected, FW NO₃⁻ concentrations mirrored NO₃⁻ flux patterns (Figure 1A; Figure 1B).

Quantification of N₂O emissions from soil surface to the atmosphere in selected treatments revealed the greatest N₂O losses in CC fields receiving either spring-applied manure or UAN (8.2 or 6.4 kg N₂O-N /ha/yr, respectively; Figure 2). Nitrogen management using fall-applied manure for CC resulted in significantly lower N₂O losses. These patterns could be explained by enhanced soil N uptake and use efficiency in CS rotation phase and by manure additions in CC during the late fall shortly prior to cold-wet soil conditions that can limit nitrification and denitrification processes (Hernandez-Ramirez *et al.* 2009b). These sharp FM vs. SM differences support the pronounced effect of time of N additions in agricultural soils on the outcomes of N₂O emissions as well as NO₃⁻ losses as discussed above. Spring UAN addition in CS rotation phase also registered numerically reduced N₂O losses perhaps due to increased soil N uptake and associated improved N use efficiency by corn. As noted above, PG system received minimum management and also soil N availability in PG is likely limited in part due to effective N competition between microbes and plant roots. These various factors can explain the negligible soil N₂O emission levels in PG (0.24 kg N₂O-N /ha/yr).

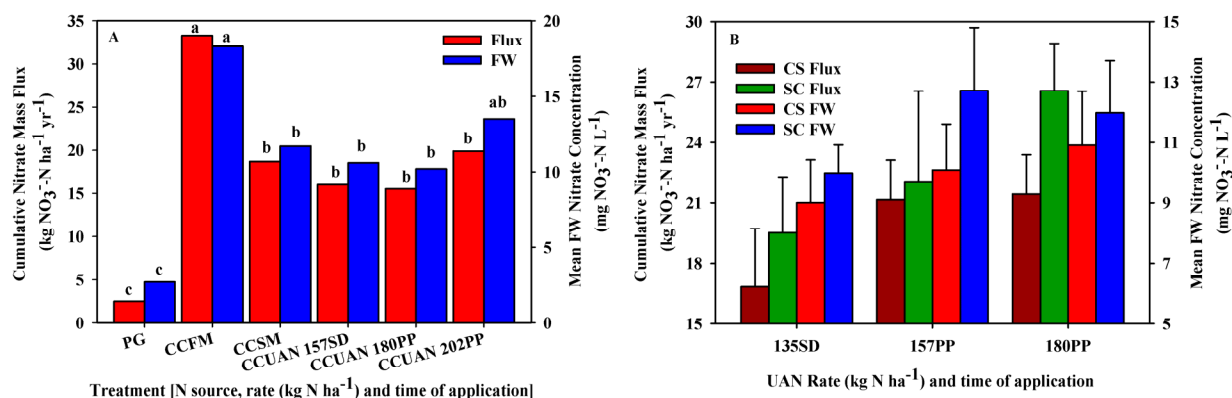


Figure 1. Annual cumulative loads and mean flow-weighted (FW) concentrations of nitrate in subsurface drainage water for (A) prairie grass (PG) and continuous corn (CC) systems, and (B) corn (CS) and soybean (SC) rotation phases. Urea-ammonium nitrate (UAN) rates and time of N additions are noted. Within each variable, treatments labelled by the same letter are not different based on Tukey's HSD test ($\alpha = 0.05$). Values are averages of six hydrological years. Error bars are SE. Note the different scales across panels.

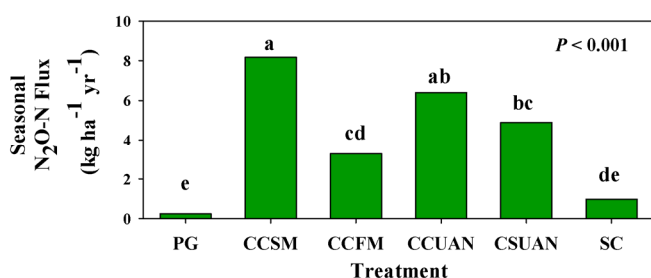


Figure 2. Annual cumulative nitrous oxide emissions (2-yr mean) at soil surface for selected treatments. Treatments labelled by the same letter are not significantly different based on Tukey's HSD test ($\alpha = 0.05$).

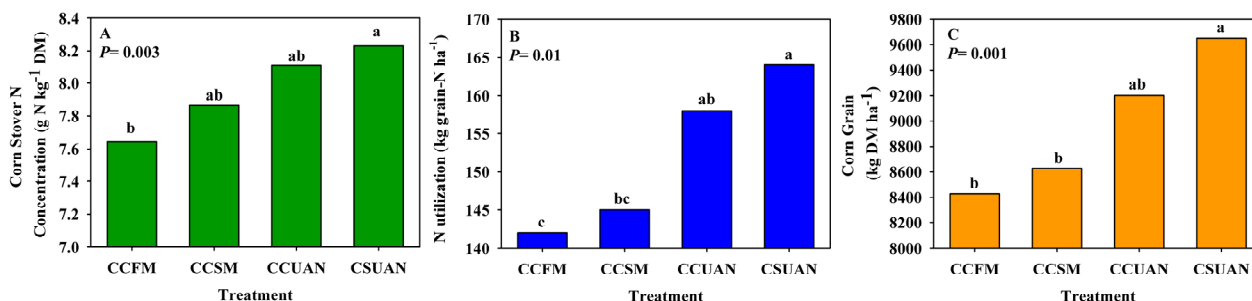


Figure 3. (A) Corn yield, (B) nitrogen utilization, and (C) stover nitrogen content for selected treatments. Treatments labelled by the same letter are not significantly different based on Tukey's HSD test ($\alpha = 0.05$).

Differential responses of soil and plant N pools to the assessed N management practices further substantiate our findings for both NO₃⁻ losses in drainage water and N₂O emissions to the atmosphere. Fractional analyses of PG soils indicate that PG soil was strongly N limited as C:N ratios were markedly wider when compared to cropping systems in both whole soil (15 vs. 13) and fine particulate organic matter (28 vs. 16; data not shown). These divergent C/N ratios between cropped and PG soils can also suggest the differential nature of soil organic matter between crop and PG systems as biomass characteristics, N inputs, and tillage management also differ (Hernandez-Ramirez *et al.* 2009a). Likewise, analyses of aboveground plant tissues across selected treatments also showed general agreement of plant N status with our observations of NO₃⁻ and N₂O losses. The N contents in corn stover trended lower in manured fields (Figure 3A), and this can be in part attributed to losses of N species to both subsurface drainage water and the atmosphere as discussed above. Moreover, results of N mass allocated into corn grain indicated even sharper differences in plant N utilization between fields receiving UAN vs. manure (Figure 3B). Relatively increased soil N uptake and associated enhanced N use efficiency by corn in fields receiving UAN could account for this outcome. The strong driving role of N management in these cropping systems is also reflected in corn productivity (Figure 3C) which in general mimicked the patterns of N contents in both stover and grain.

Ecosystem services trade offs: nitrate leaching vs. nitrous oxide emissions

Summarizing the divergent results generated by spring vs. fall manuring in corn fields, several directional trade offs in ecosystem services can be clearly identified. In contrast to CCSM, CCFM showed a beneficial ecosystem service by mitigating N₂O emissions. However, CCFM exhibited detrimental ecosystem services outcomes compared to CCSM such as reduced grain yield and much larger NO₃⁻ losses. Furthermore, as agriculturally-sourced NO₃⁻ is leached from soils and transported via drainage networks and surface watercourses, certain biological N₂O production (e.g., denitrification) typically takes place. Therefore, as a direct result of the relatively larger NO₃⁻ leaching for CCFM vs. CCSM, the estimates of indirect N₂O emissions following IPCC (2006) methodology are also greater for CCFM vs. CCSM (i.e., 0.25 vs. 0.14 kg N₂O-N /ha/yr, @ IPCC default value of 0.0075). However, these additional amounts of indirect N₂O emissions are relatively small, and they do not alter the pre-existent treatment hierarchy (i.e., CCSM > CCFM) based only on quantities of measured direct N₂O emissions.

Conclusions

This study suggests that time of N addition in corn fields had a significant driving effect on ecosystem services outcomes such as magnitudes of grain yield, N₂O emissions, and NO₃⁻ leaching. Overall ecosystem services performance could be enhanced by selecting the best time for N addition to effectively synchronize plant N demand with soil N availability as well as by implementing nutrient management plans assigning proper credits to N contributions by legume and manure additions. Fine tuning of both N budgets and fertilizer recommendations for corn production systems need to consider other components of the N cycle such as N mineralization of soil organic matter and corn residues, N release from soil mineral fraction, deep N percolation below drains, N in runoff, gaseous N losses (different than N₂O), and wet and dry N deposition. This study also indicates that PG restoration in cropland can cause beneficial environmental outcomes by minimizing effluxes of both N₂O and NO₃⁻; however, it remains unknown the potentially adverse impacts of PG biomass removal (i.e., due to growing interest in biofuel fabrication) and associated repeated N additions with the aim of maximizing biomass productivity in these PG systems. Comparative life cycle assessments for a broad variety of agroecosystems would need to encompass all known ecosystem services and environmental impacts using a common base methodology. Additional data is also needed as predictions of near-future climate changes indicate increased variability and changeable outcomes.

References

- Daily GC, Matson PA (2008) Ecosystem services: From theory to implementation. *Proc. Natl. Acad. Sci. U.S.A.* **105**, 9455-9456.
- Hernandez-Ramirez G, Brouder SM, Smith DR, Van Scoyoc GE (2009a) Carbon and Nitrogen Dynamics in an Eastern Corn Belt Soil: N Source and Rotation. *Soil Sci. Soc. Am. J.* **73**, 128-137.
- Hernandez-Ramirez G, Brouder SM, Smith DR, Van Scoyoc GE (2009b) Greenhouse Gas Fluxes in an Eastern Corn Belt Soil: Weather, N Source and Rotation. *J. Environ. Qual.* **38**, 841-854.
- Huggins DR, Randall GW, Russelle MP (2001) Subsurface drain losses of water and nitrate following conversion of perennials to row crops. *Agron. J.* **93**, 477-486.
- IPCC – Intergovernmental Panel on Climate Change (2006) Guidelines for national greenhouse gas inventories. Available at <http://www.ipcc-nggip.iges.or.jp/public/2006gl/index.html> [verified 15 Oct. 2009]. Geneva, Switzerland.
- Kaspar TC, Jaynes DB, Parkin TB, Moorman TB (2007) Rye cover crop and gamagrass strip effects on NO₃ concentration and load in tile drainage. *J. Environ. Qual.* **36**, 1503-1511.
- Palmer MA, Filoso S (2009) Environmental Markets Restoration of Ecosystem Services. *Science* **325**, 575-576.
- Tilman D, Cassman KG, Matson PA, Naylor R, Polasky S (2002) Agricultural sustainability and intensive production practices. *Nature* **418**, 671-677.
- Vitousek PM, Naylor R, Crews T, David MB, Drinkwater LE, Holland E, Johnes PJ, Katzenberger J, Martinelli LA, Matson PA, Nziguheba G, Ojima D, Palm CA, Robertson GP, Sanchez PA, Townsend AR, Zhang FS (2009) Nutrient Imbalances in Agricultural Development. *Science* **324**, 1519-1520.

Root respiration interferes with peat CO₂ emission measurement

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Abstract

Root respiration and microbial decomposition release CO₂ from peatland. Mixture between these two measurements causes an over-estimation of greenhouse gas contribution, because CO₂ produced by the former is offset by atmospheric CO₂ removal during photosynthesis. We separated the two components by measuring, from closed chambers, the CO₂ emission from the rooted (R) and non-rooted (NR) zones of peatland planted to oil palm. Three pieces of roots were channelled through a 5 cm hole into each of the grounded part of the R chamber. Emitted CO₂ was captured by 30-cm diameter and 30-cm tall PVC gas chambers, sampled using 10 ml syringes, and measured using gas chromatography. The measurements were conducted in Aceh, Sumatra in the early rainy season (Oct-Nov 2008). We found that CO₂ emitted from the NR chambers was about 62% of that of the R chamber, indicating a significant contribution of the root in producing CO₂. The average amount of emitted CO₂ from these 1, 5 and 10 year oil palm soils ranged from 18 to 24 t/ha/yr. Capturing both the R and NR zones in CO₂ emission measurement is advisable, but under limited resources, the NR zone should be prioritised.

Key Words

Peat, root respiration, microbial decomposition, CO₂ emissions

Introduction

Gas capture from closed chambers is one of the most common techniques in green house gas emission studies. CO₂ flux from peat soil as measured from the closed chamber is produced by both heterotrophic (microbial) and autotrophic (root) respiration. While the former component is related to peat decomposition and thus a net CO₂ emission, the latter is a near carbon neutral process; i.e. CO₂ is removed from the atmosphere though photosynthesis and perhaps lesser amount of CO₂ is released from root respiration. As such, measured CO₂ emission from the soil could be over-estimated if these two components are not separated. This research was aimed at assessing the effect of the rhizosphere on CO₂ emissions. The level of root respiration influence can be used to correct the figure of peat decomposition related oxidation.

Methods

This research was conducted in the peat domes in three villages of Aceh Barat District, Nanggroe Aceh Darussalam (NAD) Province of Indonesia in Nov. to Oct. 2008. The emitted CO₂ gas was captured using closed chambers from the rooted (R) and non-rooted (NR) chambers or zones.

The observation was conducted at smallholder oil palm plantations with 10, 5, and 1 year old plant stands. For each of the age group five pairs of observation points were made; each with R and NR treatment. The observation points of the five pair transect were arranged at 10, 55, 100, 145 and 190 m perpendicular to the drainage canal. The drainage canal where the measurement were taken, varied in depth between 1.5 m to 0.7 m depending on the position of observation points relative to tertiary canal or shallower field canal. As the water table level in the canal fluctuates in the canal as well as measurement points, water table depth was measured using a measuring stick from auger hole as deep as 1 m at the midpoint between the R and NR. The closed chambers of 30 cm diameter and 30 cm tall were made of PVC tubes. The bottom brim of the chamber was sharpened to minimize soil compaction during their insertion into the ground.

For each chamber designed for rooted zone emission, a hole of 5 cm diameter at a point 20 cm from its top was made for channelling three pieces of oil palm roots in such a way that the roots can still grow and develop inside the chambers. These chambers were installed at a distance of 2.5 m, 1 m and <1 m from the trunk of oil palm aged 10 years, 5 years and 1 year, respectively according to the distribution of the plant roots. The paired R and NR chambers were mounted at a distance of 1 m from each other.

Each chamber was equipped with a septum to place the needle puncture. A small (6 cm battery powered) fan was installed inside the chambers to stir the gas. A thermometer was also installed for each chamber to measure the temperature during the gas sampling.

Gas samples were taken by using syringes 10 ml capacity, with a sampling frequency of 0, 5, 10, 15, 25 and 35 minutes after closing of the chambers. Sampling was conducted at 07:00 to 10:00 a.m. and emission during this time range is assumed to represent the emission during the 24 hours. Samples were analyzed within 24 hours after sampling using a portable gas chromatography. The difference between R and NR treatments were tested using the t-test pair-wise comparison.

Results

The pair-wise comparison test shows that CO₂ emission from the rhizosphere zone is significantly higher than that of non rhizosphere ($p = 0.000$; Table 1). Under this experiment condition the non-rhizosphere CO₂ emission was about 62% of that of the rhizosphere one. This means that the measurement of CO₂ emission made on the rhizosphere zone overestimates the assessment of CO₂ emission, because the root respiration does not contribute to the net CO₂ emission as it is offset by CO₂ removal during photosynthesis. Where the NR data are available, they are more reliable for the reflection of peat decomposition in the green house gas studies. Our data of non rhizosphere emissions, with mean values ranging from 18 to 24 t CO₂/ha/yr are less than half than the predicted values of Hooijer *et al.* (2006) of around 54 t CO₂/ha/yr under the drainage depth (in this case, the water table) of around 60 cm, Melling *et al.* (2005) of around 50 t CO₂/ha/yr for oil palm in Sarawak Malaysia and Jauhiainen *et al.* (2001) for stabilized agriculture on peatland.

For each age group, CO₂ emission under the R zone is consistently higher than the NR zone, although for the 1-year age group the p-value is rather high ($p > 0.20$). This is presumably because of the limited measurements for this age group. The amount of CO₂ emission from the 1-year old oil palm is higher than that of the older palm (Table 1), and Jauhiainen (2001) attributed this to the relatively higher content of raw organic matter that are more readily decomposed as is the case in the young plantation relative to the older ones.

Table 1. Mean \pm standard deviation and the t-test for pair-wise comparison between the non rhizosphere (NR) and the rhizosphere (R) treatments of CO₂ flux at different age of oil palm.

Age	NR	R	N	t-test
Year	t CO ₂ /ha/yr			
1	24.3 \pm 9.7	40.9 \pm 18.0	8	0.2109
5	18.2 \pm 11.1	27.3 \pm 15.6	27	0.0001
10	19.3 \pm 16.6	32.9 \pm 20.7	21	0.0020
Average	19.5 \pm 13.2	31.3 \pm 18.3	56	0

Hooijer *et al.* (2006), from a literature review, presented a positive linear correlation between CO₂ emission and water table depth. Likewise, our unpublished observation in West Kalimantan Province showed lower carbon stocks at points closer to the deep drainage canal, indicating a more rapid decomposition at these points where water contents are likely lower than those points further away from the canal.

In this current research, however, the effect of instantaneous water table depth on CO₂ emission was rather inconsistent (Figure 1). Jauhiainen *et al.* (2008) demonstrated that the highest rate of emissions were at points where water table depth is around 60 cm. When the water table depth is shallower, then the soil water content tended to be more saturated, whereas, when the water table is deeper, the soil water content is dry and thus less favourable for microbial activities.

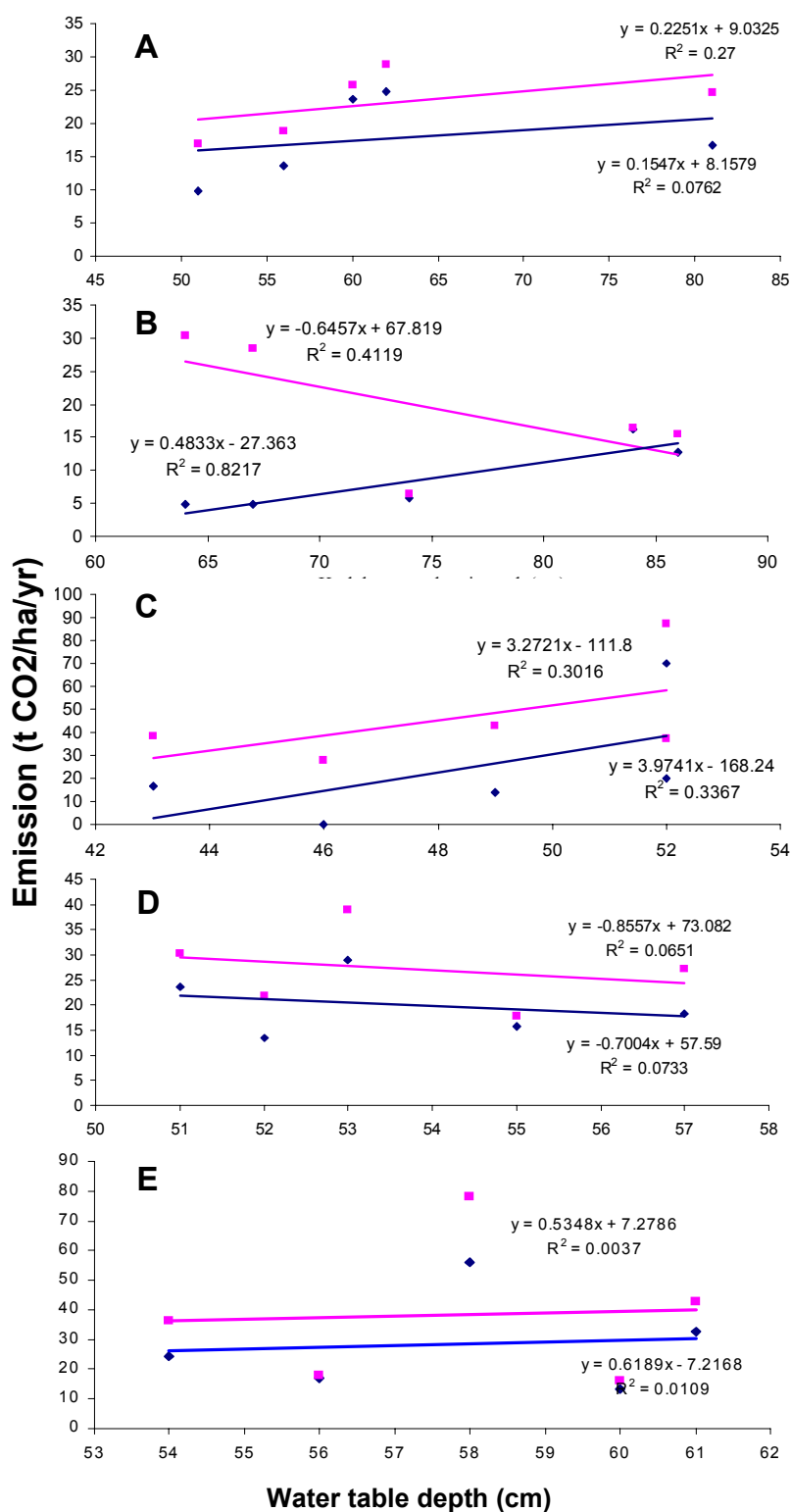


Figure 1. CO₂ flux under smallholder oil palm plantation in transects: **A.** Under 10 year palm stand, 1.5 m drainage depth; **B.** Under 10 year palm stand, 1.5 m drainage depth; **C.** Under 10 year palm stand, 0.7 m drainage depth; **D.** Under 5 year palm stand, 0.7 m drainage depth; **E.** Under 5 year palm stand, 0.7 m drainage depth. The red line is rooted and blue line is non-rooted zones.

Conclusions

Our research shows that it would cause a strong overestimate of peat soil contribution to CO₂ emission when the measurement is conducted in rooted area. Thus when zoning of the measurement is not possible, measurement should be conducted on relatively root free areas.

References

- Hooijer A, Silvius M, Wösten H, Page S (2006) PEAT-CO₂, Assessment of CO₂ emissions from drained peatlands in SE Asia. Wageningen: Delft Hydraulics report Q3943.
- Jauhiainen J, Heikkinen J, Martikainen PJ, Vasander H (2001) CO₂ and CH₄ fluxes in pristine peatswamp forest and peatland converted to agriculture in central Kalimantan, Indonesia. *International Peat Journal*. **11**, 43-49.
- Jauhiainen J, Limin S, Silvennoinen H, Vasander A (2008) Carbon dioxide and methane fluxes in drained tropical peat before and after hydrological restoration. *Ecology* **89**(12), 3503-3514.
- Melling L, Hatano R, Goh KJ (2005) Soil CO₂ flux from three ecosystems in tropical peatland of Sarawak, Malaysia. *Tellus* **57B**, 1-11.

Soil ecosystem services in Amazonian pioneer fronts: Searching for socioeconomic, landscape and biodiversity determinants

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Abstract

In two Amazonian regions of Brazil and Colombia that represent most of the diversity of the pioneer front landscapes, we searched for relationships among socioeconomic environments, landscape composition and structure, biodiversity, and production of goods and ecosystem services. An original sampling protocol was applied to collect fully compatible socioeconomic, landscape, agronomic and ecological datasets allowing rigorous statistical analyses. In each country, 153 farms belonging to three different kinds of land use and practices were characterized on the basis of socioeconomic and landscape variables. Biodiversity, goods and ecosystem services were measured on a selection of 27 (26 in Colombia) farms most representative of the whole diversity in each country. Among the groups chosen for biodiversity survey, plants, earthworms, termites and ants were major ecosystem engineers that play a critical role in the provision of goods (agrosilvipastoral products) and ecosystem services (ES). The investigated ES were climate regulation through carbon sequestration in soil and biomass, soil conservation and water cycle regulation through infiltration, and finally indices of soil quality. Co-variations among the different sets of variables assessed by multiple co-inertia analysis were highly significant. Significance of these results are discussed.

Key Words

Land use change, socioeconomic drivers, carbon sequestration, soil quality, Brazil, Colombia.

Introduction

Amazonian pioneer fronts are highly dynamic areas where deforestation occurs at a very high rate, although in rather diverse ways since land use may vary considerably depending on the socioeconomic environment, local geographical conditions, biodiversity and land use strategies (Fearnside 2005).

Pioneer agriculture, right after deforestation, is often based on very simple slash and burn systems. They have negative effects on soil quality and result in a general decrease in the production of soil ecosystem services. A large number of studies show significant decreases in climate regulation processes through depletion in carbon and decreases in plant biomass (Nepstad *et al.* 2008, Kauffman *et al.* 2009). The ability of soil to sustain primary production is also endangered through a variety of mechanisms, nutrient depletion being the most conspicuous one. Water resource services are also impaired through decreased infiltration, storage and transfers of water as soils get compacted in pastures and suffer erosion (Zimmermann *et al.* 2006). Biodiversity is severely decreased, especially that of soil ecosystem engineers (earthworms, termites and ants), although with relatively diverse patterns (Mathieu *et al.* 2005). However, some specific practices e.g., cropping systems with perennial tree productions, may be less detrimental than others. Furthermore, landscape composition and structure, i.e, the composition of the mosaic of different types of land uses that people create from the original forest- is likely to influence the whole dynamics (Barros *et al.* 2002).

The present study assesses the relationships among the production by soils of goods and ecosystem services, socio economic parameters, landscape composition and structure, biodiversity in two rather contrasted regions, an area of relatively recent colonization (10-15 years) in North Eastern Brazil (state of Pará) and a region colonised 60 years ago in South West Colombia (Caqueta). We tested the hypothesis that socio economic levers amenable for changes via public policies are key determinants of a suite of interactions that determine soil functions. The construction of different landscapes that host different biodiversities eventually affects parameters of soil functions.

Methods

In each country, three landscape “windows” each formed by 3 replicate groups of 17 contiguous farms were selected with different ages of deforestation and/or different land tenure (Table 1).

Table 1. Main characteristics of the surveyed landscape “windows”

Country	Landscape “windows”	Beginning of the deforestation	Average area of the exploitations (ha)	% forest
BRAZIL	Palmares II	1990	25	44
	Maçaranduba	1994	60	40
	Pacajá	1997	60	70
COLOMBIA	Traditional	1950	64	2
	Agrosilvipastoral	1940	20	2
	Agroforestry	1950	21	6

Socio-economic characterisation was performed on 153 farms in each country with three different questionnaires that addressed respectively individual life histories (32 variables describing migrations, studies, professional abilities, family), economic situation (15 variables describing different kinds of incomes and access to credits) and production systems (21 variables).

Landscape analysis done after maps of the 28 different types of land use identified allowed to quantify landscape composition (amount and % of the different types of land use) and structure (patchiness, diversity, distance among patches of a similar type of habitat) using FRAGSTAT program.

In a selection of 53 farms representing the diversity of socioeconomic situations in each of the 6 landscape windows, detailed studies of biodiversity and soil ecosystem services were performed.

- Biodiversity of plants, soil invertebrate engineers (termites, earthworms, ants, and general soil macro invertebrate communities), birds, moths (Saturnidae, Sphingidae), Drosophilidae and bees was assessed in each of the 53 farms, at five points regularly spaced 200 m apart along a line located in the middle of the farm. Main types of land use in each farm were thus rather well represented in our sampling.
- Production of forest, agriculture and cattle breeding activities were measured accurately in each farm and expressed either as amounts produced or caloric and protein equivalents.
- Soil attributes were thoroughly measured through physical, chemical, organic matter and morphology characteristics. They were then grouped by categories and indices of soil quality were calculated according to the GISQ methodology (Velasquez *et al.* 2007a).

Once obtained entirely compatible data tables, co-inertia analysis (Dolédéc and Chessel 1994) were performed among each pair of tables to test for significant co-variations (i.e., correlations among tables measured as the as the vectorial correlation and noted RV, Robert and Escoufier 1976).

Results

Co-inertia analyses among the 12 different tables provided a large number of significant relationships (Figure 1). The hypothesis of cascading effects from socioeconomic to landscape, biodiversity and the production of services and agrosilvipastoral goods was thus validated.

	WP4_PROD	WP4_PRKC	WP1_SYPRO	WP1_SOCF	WP1_SOCQ	WP2_STRU	WP2_COMP	WP2_USA	WP3_BIODIV	WP4_MORP	WP4_PHCH
WP4_PRKC	0.37										
WP1_SYPRO	0.48	0.23									
WP1_SOCF	0.23	0.18	0.42								
WP1_SOCQ	0.35	0.18	0.38	0.47							
WP2_STRU	0.35	<i>0.09</i>	0.29	0.35	0.26						
WP2_COMP	0.30	0.26	0.27	0.27	0.31	0.27					
WP2_USA	0.49	0.24	0.33	0.36	0.30	0.18	0.20				
WP3_BIODIV	0.43	<i>0.13</i>	0.38	0.46	0.46	0.29	0.28	0.52			
WP4_MORP	0.33	0.19	0.24	0.34	0.29	0.18	0.25	0.38	0.44		
WP4_PHCH	0.40	<i>0.10</i>	0.36	0.50	0.40	0.23	0.24	0.47	0.69	0.52	
WP4_GISQ	0.38	0.17	0.28	0.19	0.35	0.25	0.24	0.41	0.49	0.48	0.58

Figure 1. Matrix of RV coefficients among the 12 tables of data obtained for the Colombian and Brazilian sites. In bold, permutation tests (n=999) significant (with $p < 0.01$ (often < 0.001); in italics, tests with a $p < 0.05$ significance.

WP4_PRKC: Agrosilvipastoral productions expressed in Kcal and amount of glucids, lipids and proteins produced; WP1_SYPRO: production systems; WP1_SOCF: life histories and other social information; WP1_SOCQ: economic information; WP2_STRU: landscape structure in a 100m radius circle around sampling point; WP2_COMP: landscape composition; WP2_USA: land uses at the sampling point; WP3_BIODIV: biodiversity of plants and 7 groups of animals; WP4_MOR: soil morphology as assessed by the Velasquez *et al.* (2007b) method. WP4_PHCH: soil physicochemical variables; WP4_GISQ: indicators of soil quality.

The main results regarding determinants of soil ecosystem services were as follows:

1. There was a significant effect of socioeconomic parameters over land use intensification. People living closer to cities, or with good quality infrastructures, would generally get a better access to education and credit and have more intensified practices. Under such conditions, less C is stored into plant biomass and soil degradation tends to be higher, especially when livestock breeding is the main activity.
2. Chemical fertility was higher in all derived systems in Brazil due to the incorporation of ashes in soil; differences among sites were mainly due to differences in soil texture. In Colombia, acidic soils with high Al saturation and lower base contents were found thus showing that the initial correction of pH observed in Brazil may no longer persist 30 to 40 years after deforestation. However, comparison among sites show differences in Colombia, since silvipastoral and agroforestry systems respectively show improved conditions as compared to the traditional system.
3. Soil C storage is greatly dependent on clay concentrations and soil depth. As a result, Colombian soils have higher C contents in general; in Brazil, the Palmares site that has significantly higher clay contents and soil depths than the other two, also stores more C. Soil C storage was not significantly affected by land use types (neither in Brazil nor in Colombia).
4. Soil aggregation was greatly influenced by clay contents in Brazil. However land use types also had effects and a larger proportion of biogenic aggregates were found in improved systems in Colombia as compared to the traditional system.
5. Compaction was generally observed in pasture sites as compared to other types of land use. In Brazil, field measurements showed a clear correlation between bulk density values and infiltration rates. In pastures, infiltration rates were 10 times slower in pastures than in adjacent forests.
6. Soil macrofauna was greatly affected by land use types. While no difference was observed at the order level among Brazilian sites, a longer period of use resulted in significant decrease in Colombia, with the greatest impact measured in the intensive livestock traditional site (CTR) where continuous grazing of degraded pastures in a largely deforested areas severely decreased the diversity, if not the abundance, of macroinvertebrate communities (Figure 2). Termites were the most affected group while earthworms suffered shifts in community composition with a very high predominance of the invasive species *Pontoscolex corethrurus*.

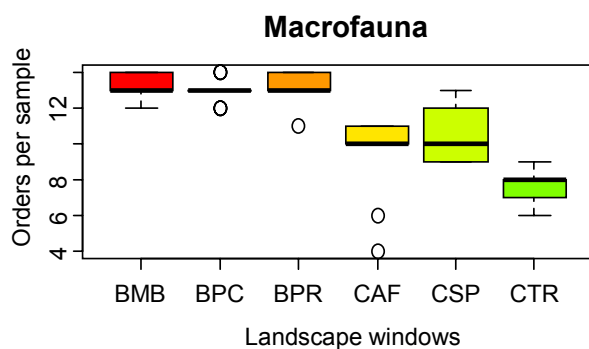


Figure 2. Variation of soil macrofauna diversity (order level) among surveyed sites of Brazil (BMB: Maçaranduba, BPC: Pacajá, BPR: Palmares II) and Colombia (CAF: Agroforestry, CSP: Agrosilvipastoril, CTR: Traditional)

Conclusion

Land use types and their distribution in Amazonian landscapes profoundly affect soil chemical, physical and biological parameters. Deforestation and conversion to pasture or cropland degrade all parameters of soil quality, and hence the production of ecosystem services. Degradation, however, proceeds at different rates according to the production system implemented and also to time elapsed since first deforestation. Systems that maintain trees (e.g., extractivist exploitation or agroforestry systems) have less detrimental effects. On the other hand recently deforested areas of Brazil seemed to have kept better abilities for production of ecosystem services than Colombian systems deforested since a much longer time. Our study also revealed a particular importance of landscape composition and structure, showing that intensive systems when limited in area and associated to more conservative systems in a diverse landscape mosaic may have less detrimental effects.

References

- Barros E, Pashanasi B, Constantino R, Lavelle P (2002) Effects of land-use system on the soil macrofauna in western Brazilian Amazonia. *Biology and Fertility of Soils* **35**, 338–347.
- Dolédec S, Chessel D (1994) Co-inertia analysis: an alternative method for studying species-environment relationships. *Freshwater Biology* **31**, 277–294.
- Fearnside PM (2005) Deforestation in Brazilian Amazonia: history, rates, and consequences. *Conservation Biology* **19**(3), 680–688.
- Kauffman JB, Hughes RF, Heider C (2009) Carbon pool and biomass dynamics associated with deforestation, land use, and agricultural abandonment in the neotropics. *Ecological Applications* **19**(5), 1211–1222.
- Mathieu J, Rossi JP, Mora P, Lavelle P, Martins PFS, Rouland C, Grimaldi M (2005) Recovery of soil macrofauna communities after forest clearance in Eastern Amazonia, Brazil. *Conservation Biology* **19**, 1598–1605.
- Nepstad DC, Stickler CM, Filho BS, Merry F (2008) Interactions among Amazon land use, forests and climate: prospects for a near-term forest tipping point. *Phil. Trans. R. Soc. B* **363**, 1737–1746.
- Robert P, Escoufier Y (1976) A unifying tool for linear statistical methods: the RV-coefficient. *Applied Statistics* **25**, 257–265.
- Velasquez E, Lavelle P, Andrade M (2007a) GISQ, a multifunctional indicator of soil quality. *Soil Biology and Biochemistry* **39**, 3066–3080.
- Velasquez E, Lavelle P, Andrade M (2007a) GISQ, a multifunctional indicator of soil quality. *Soil Biology and Biochemistry* **39**, 3066–3080.
- Velasquez E, Pelosi C, Brunet D, Grimaldi M, Martins M, Rendeiro AC, Barrios E, Lavelle P (2007) This ped is my ped: Visual separation and near infrared spectra allow determination of the origins of soil macroaggregates. *Pedobiologia* **51**, 75–87.
- Zimmerman B, Elsenbeer H, Moraes JM (2006) The influence of land-use changes on soil hydraulic properties: Implications for runoff generation. *Forest Ecology and Management* **222**, 29–38.

Soil erosion potential under forest vegetation in the humid subtropics of southeast China

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Abstract

Sand-filled splash cups were used to study the erosive power of rainfall and throughfall in the humid subtropics of southeast China. Our results showed that the splash cup measurements yielded precise and reproducible results both under open field conditions and under forest vegetation. The splash cups were exposed to forest stands of different age and to selected species (*Schima superba*, *Castanopsis eyrei*, *Daphniphyllum oldhamii*, *Lithocarpus glaber*) in the Gutianshan National Nature Reserve (GNNR). The results obtained under forest vegetation show that the erosive power of throughfall drops is about 2.59 times higher compared to the open field, which accentuates the importance of shrub, herb and litter layers in forest ecosystems to protect the soil against erosion. Coalescing drops from leaves and branches (drips) are responsible for this enormous gain in erosive power. Further, differences in sand loss between the investigated tree species (deciduous, evergreen) showed that the erosion potential and the spatial heterogeneity of throughfall is species-specific, highlighting the importance of selecting specific species for afforestation projects considering soil erosion potential.

Key Words

BEF-China, biodiversity, soil erosion, soil erosion experiment, splash cups, throughfall erosivity.

Introduction

In soil erosion research it is widely accepted that vegetation is a key control for the type and intensity of erosion (e.g. Hudson 1971; Thornes 1990; Morgan 2005). The current paradigm is that natural or quasi-natural vegetation protects the soil against erosion while agricultural land use generally enhances erosion. But even severe soil erosion may take place under forest vegetation with a well developed canopy but high amounts of bare ground due to lack of a shrub or herb layer (Zhao 2006; Nanko *et al.* 2008). Especially on steep slopes relocation of litter results in a patchy distribution of plant residues and litter. Thus parts of the forest floor remain uncovered and unprotected (Tsukamoto 1991). Moreover climate change induces a shift of the precipitation regime to exceptional intense rainfall events. Key mechanisms of a vegetation cover in reducing or enhancing erosion potential are the modification of drop size distribution, retention of raindrop impact on the soil and changes in amount and spatial distribution of rainfall at the ground surface. This study focuses on the application of sand-filled splash cups to study rainfall and throughfall erosivity in natural systems. We will show the relationship of rainfall and vegetation characteristics to the sand loss measured by splash cups.

Materials and methods

This study was conducted in the Gutianshan National Nature Reserve (GNNR), Zhejiang Province, P.R. China. The GNNR is located at N 29°14.657' and E 118°06.805' (center). The elevation ranges between 320 m and 910 m above sea level. The soils are predominantly Cambisols (cf. IUSS Working Group WRB 2007) developed on granite with a more or less thick saprolite cover. The climate at the GNNR is typical of subtropical monsoon regions with an annual average temperature of 15.3°C and a mean annual rainfall of 1963.7 mm (Hu and Yu 2008). Rainfall data was obtained from an automatic weather station in the centre of the GNNR.

Modified splash cups

The modified splash cups have a diameter of 4.6 cm and a surface of 16.619 cm². They were developed based on the archetype of Ellison (1947). We measure the unit sand remaining inside the cup after single natural rainfall events. By using unit sand with distinct properties (grain size: 125-200µm) it is possible to focus on other factors such as the differing characteristics of open rainfall and throughfall (Salles and Poesen 2000) or wind effects to rain (Cornelis *et al.* 2004). Sandloss as the target variable is calculated as the difference of initial weight and resulting weight. For a detailed description and calibration results of the splash cups, see Scholten *et al.* (submitted) and Geißler *et al.* (submitted).

Experimental design

The experimental design of the splash cup measurements under vegetation consists mainly of two parts: forest stand-based and species-based. The specific forest stands were selected along a biodiversity and succession gradient. To calibrate the method a set of five splash cups (surface = 83.095 cm²) was positioned in a pentagon next to the climate station under open field conditions. To reveal effects of single tree species on the erosivity three sets of five splash cups at a time were positioned under the target trees. A target tree represents a typical individual of one of the selected species (*Schima superba*, *Castanopsis eyrei*, *Daphniphyllum oldhamii*, *Lithocarpus glaber*). The splash cups were established 1 m above ground to avoid disturbance by animals, forest floor vegetation and differing inclination of slopes. One rainfall collector (collecting area = 13.396 cm²) was positioned per splash cup. An outlier test was applied both to the rainfall data and the splash cup measurements with standard deviation * 2 + mean as a criterion for exclusion.

Results and discussion

Simultaneous measurements under open field conditions and in forest stands of the three successional stages showed a remarkable difference of sand loss (Figure 1). Throughfall in all varies between 57% and 70% of total precipitation. The average sand loss under vegetation (44.69 g/m²) was about 2.59 times higher than under open field conditions (17.24 g/m²). There was a non-linear increasing sand loss with increasing successional stage: the difference between the young successional stage and the intermediate successional stage (16.71 g/m², factor 1.51) is much higher than between the intermediate successional stage and the old successional stage (1.45 g/m², factor 1.03)(Figure 2).



Figure 1. Splash cup under open field conditions (a), under forest vegetation (b).

The resulting standard deviations between the measurement replicates can be considered in two ways: (i) For open field conditions the standard deviation is much lower than for all successional stages and (ii) the difference in standard deviation between the young successional stage (average 2.29) and the intermediate successional stage (average 3.61) was much higher than between the intermediate and the old successional stage (average 3.82). The standard deviation found under two common species (*Schima superba*, *Castanopsis eyrei*) was much higher (average 21.12 and 34.78 respectively) than under open field conditions (average 0.84). Nevertheless, there were significant differences in splash potential represented by sand loss between *Schima superba* (55.54 g/m²) and *Castanopsis eyrei* (42.18 g/m²), although the average amount of throughfall was quite similar (24.77 mm for *Schima superba* and 23.22 mm for *Castanopsis eyrei*). The erosion potential of throughfall drops in forests was considerably higher (2.59 times) than under open field conditions and the spatial variability of the erosion potential was much more diverse in forests, caused by free throughfall and drip. By placing the splash cups under certain tree species it could be demonstrated that the erosivity of throughfall depended on the species and also on the successional stage of the forest stand.

Schima superba for example generated throughfall drops of higher erosivity than *Castanopsis eyrei*. We assume that this is an effect of the larger size of the leaves which generates larger throughfall drops, particularly drip. The erosion potential is also a function of stand height because throughfall drops are much more likely to reach terminal velocity under old grown forests. Therefore we conclude that the potential of throughfall to detach soil by splash is also related to the age of the specific forest stand: young forests have the lowest and old forests the highest erosion potential. This accentuates the importance of a shrub, herb and litter layer in forest ecosystems to protect the soil against erosion.

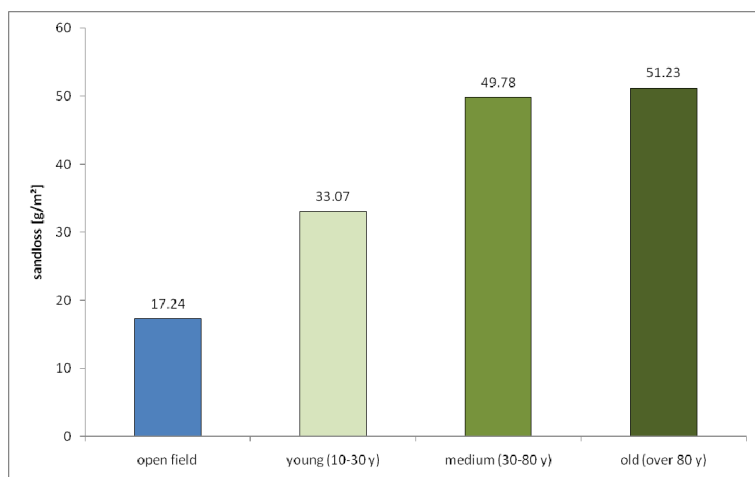


Figure 2. Throughfall erosivity as measured by splash cups under open field conditions and three successional stages.

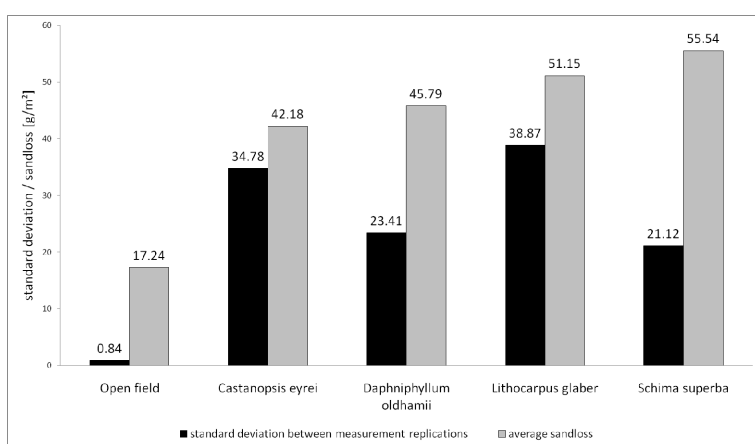


Figure 3. Erosive power of throughfall as measured by splash cups for four tree species.

Conclusion

We studied rainfall and throughfall erosivity in a subtropical forest ecosystem using sand-filled splash cups. Our results showed that the splash cup measurements yielded precise and reproducible results under both open field conditions and forest vegetation. The erosion potential of throughfall drops in forests was considerably higher (2.59 times) than under open field conditions and the spatial variability of the erosion potential was much more diverse in forests, caused by free throughfall and drip. By placing the splash cups under certain tree species and specific forest stands it could be demonstrated that the erosivity of throughfall depended on the species and also on the successional stage. *Schima superba* generated throughfall drops of higher erosivity than *Castanopsis eyrei*. We assume that this is an effect of the larger size of the leaves which generates larger throughfall drops, particularly drip. The erosion potential is also a function of stand height because throughfall drops are much more likely to reach terminal velocity under old grown forests. Therefore we conclude that the potential of throughfall to detach soil by splash is also related to the age of the specific forest stand: young forests have the lowest and old forests the highest erosion potential. This accentuates the importance of a shrub, herb and litter layer in forest ecosystems to protect the soil against erosion.

References

- Cornelis WM, Oltenfreiter G, Gabriels D, Hartman R (2004) Splash-Saltation due to Wind-Driven Rain: Vertical Deposition Flux and Sediment Transport Rate. *Soil Science Society of America Journal* **68**, 32–40.
- Ellison WD (1947) Soil Erosion Studies – Part II. *Agricultural Engineering* **28**, 197–201.
- Geißler C, Kühn P, Böhnke M, Bruehlheide H, Shi X, Scholten T (2010) Measuring splash erosion potential under vegetation using sand-filled splash cups. *Catena* (submitted)
- Hudson N (1971) 'Soil Conservation'. (Batsford: London).

- Morgan RPC (2005) 'Soil erosion and conservation'. 3rd edition. (Blackwell: Malden, MA).
- Nanko K, Mizugaki S, Onda Y (2008) Estimation of soil splash detachment rates on the forest floor of an unmanaged Japanese cypress plantation based on field measurements of throughfall drop sizes and velocities. *Catena* **72**, 348–361.
- Salles C, Poesen J (2000) Rain properties controlling soil splash detachment. *Hydrological Processes* **14**, 271–282.
- Scholten T, Geißler C, Wiegand C, Goc J, Kühn P (2010) A new splash cup to measure the erosion potential of rainfall under vegetation. (submitted)
- Thornes JB (1990) 'Vegetation and erosion – processes and environments'. (Wiley: Chichester).
- Tsukamoto J (1991) Downhill movement of litter and its implication for ecological studies in three types of forest in Japan. *Ecological Research* **9**, 333–345.
- Zhao QG (2006) Some Considerations for Present Soil and Water Conservation and Ecology Security of South China. *Bulletin of Soil and Water Conservation* **26**, 1–8. (in Chinese)

Spatial isolation increases diversity of complex bacterial community

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Abstract

One of soil ecology's most intriguing puzzles is how so many different bacterial species can coexist in small volumes of soil. This study tested the theory of spatial isolation which proposes that in unsaturated soils with high clay content, bacterial diversity is greater because fragmented aqueous habitats and thin water films spatially isolate bacterial communities from each other. The texture of a sandy soil was altered without altering chemical properties by adding 0 or 10% silt- and clay-sized particles. Both textures were incubated at several water contents. The effect on the active bacterial communities was measured using terminal restriction fragment length polymorphism (TRFLP) of bacterial 16S rRNA. Bacterial diversity was higher at the lowest soil water content ($P < 0.012$), but was not affected by texture ($P > 0.553$). Bacterial community structure in soil was affected by water content and texture ($P < 0.001$). In ordination plots, bacterial communities were separated into those incubated at water contents of $<$ or $> 56\%$ water-filled pore space and those in the two textures. This study supports the hypothesis that water content affects the spatial isolation of complex bacterial communities in soil. Given that most soils are unsaturated, these findings suggest that spatial isolation may provide a fundamental theory to explain the high diversity of bacteria in soil.

Key Words

Bacterial diversity, soil, spatial isolation, soil ecology, TRFLP.

Introduction

Soil bacteria are among the most diverse groups of organisms on earth (Torsvik *et al.* 2002) and 4×10^6 taxa may coexist in a single tonne of soil (Curtis *et al.* 2002). One of soil ecology's most intriguing puzzles is how so many different bacterial species can coexist when competition theory predicts that less competitive species would decline and eventually disappear (Tilman 1982; Deschesne *et al.* 2007). Tiedje *et al.* (2001) proposed the theory of spatial isolation which predicts that in unsaturated soils with high clay content, fragmented aqueous habitats and thin, tortuous water films spatially isolate bacterial communities from each other (Tiedje *et al.* 2001). These conditions may reduce competition and allow bacterial species to coexist. Low water contents and slow rates of diffusion have been shown to favour the coexistence of two bacterial species competing for a single substrate in modelling studies (Long and Or 2005; Dens and Van Impe 2001) and laboratory studies in agar (Dechesne *et al.* 2008) and artificial soils (Treves *et al.* 2004). There is relatively little evidence about the effect of soil texture on bacterial diversity, but it has been shown to be higher in the clay fraction of soil than the sand fraction (Ranjard *et al.* 2000; Sessitsch *et al.* 2001; Torsvik and Ovreas 2002). The theory of spatial isolation has not been experimentally tested in a field soil with a complex bacterial community and a variety of naturally occurring substrates. In this study we tested hypothesis that the spatial isolation contributes to the high diversity of bacterial communities in soil.

Methods

The texture of a quartz-based sandy soil was altered by adding 0 or 10% silt-and clay-sized particles of quartz (Rocla Quarry Products, Australia). Each texture ('sand' and 'silt+clay') was incubated for 7 d at water potentials between -15 cm (high water content) and -55 cm (low water content). The effect of water potential and texture on the diversity and structure of bacterial communities was measured using TRFLP of bacterial 16S rRNA.

Results

Water potential and texture significantly affected bacterial community structure ($P < 0.001$). In ordination plots, bacterial communities clustered into four groups depending on whether they were incubated in sand or silt+clay and whether the water-filled pore space of soil was $< 56\%$ or $> 56\%$ (Fig. 1). Most pairwise comparisons between bacterial communities in different clusters were significantly different ($P < 0.05$). Water potential significantly affected the richness (S , eH') and diversity (H' , $1/D$) of bacterial communities ($P < 0.012$) but texture did not ($P > 0.553$). In both textures, the richness and diversity of bacterial communities was higher at water-filled pore space $< 56\%$ than at water-filled pore space $> 56\%$ (Table 1).

Table 1. The richness, evenness & diversity of bacterial RNA-TRFLP profiles incubated in 0% silt+clay (sand) and 10% silt+clay (Silt+clay) at a range of water potentials. Least significant difference (lsd, $P < 0.05$) was determined by two-way analysis of variance. Mean (\pm standard error), TRFLP profiles $n = 4$.

Texture	Water potential	Bacterial TRFLP profiles				
		S	eH'	H'	1/D	J'
Sand	15	105	58	4.05	43	0.871
	20	102	56	4.03	40	0.870
	25	116	73	4.28	54	0.901
	30	115	66	4.18	47	0.882
	40	112	66	4.19	47	0.888
	55	119	71	4.26	50	0.891
Silt+clay	15	101	60	4.09	46	0.886
	25	100	56	4.03	42	0.875
	35	107	65	4.17	50	0.892
	40	111	70	4.24	54	0.901
	45	111	71	4.25	55	0.903
	55	118	70	4.24	50	0.890
lsd		11	14	0.23	13	0.036

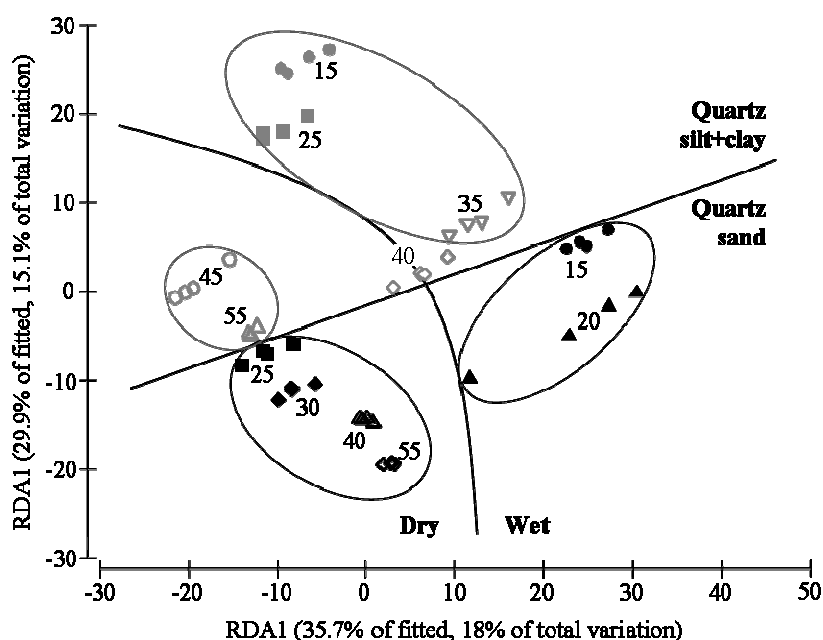


Figure 1. Distance-based redundancy analysis (dbRDA) of bacterial communities incubated at water potentials shown (-cm) in either sand (black symbols) or silt+clay (grey symbols, separated by straight line). Circles surround bacterial communities that did not differ from each other ($P > 0.05$). The curved line separates bacterial communities incubated in 'Wet' and 'Dry' soils (water-filled pore space $>$ or $<$ 56%). All pairwise comparisons across the curved line were significantly different ($P < 0.05$, except silt+clay -15 cm and -55 cm which did not differ). Soil variables used to generate the dbRDA were: water-filled pore space (% total pore volume), silt+clay content (% wt/wt), largest water-filled pore and the volume of water in pore size classes 68-77, 77-88, 88-102, 102-122, 122-153 and 153-204 μm . The figures in brackets indicate the percentage of the fitted and total variation explained by each axis.

Conclusion

The findings of this study support the hypothesis that bacterial diversity in soil increases as water content decreases. Previous studies testing the theory of spatial isolation have used only two bacterial species competing for a single substrate and were either modelling studies (Long and Or 2005; Dens and Van Impe 2001) or did not use field soils (Dechesne *et al.* 2008; Treves *et al.* 2004). In contrast, the present study used a molecular technique (RNA-TRFLP) to show that decreasing the water content of soil increased the diversity and altered the structure of a complex, indigenous bacterial community in a field soil with a variety

of naturally occurring substrates. Since field soils are usually unsaturated, the theory of spatial isolation may provide a fundamental principle to explain the high diversity of bacteria in soil (Tiedje *et al.* 2001). Although increasing the silt+clay content was not shown to increase bacterial diversity, it did alter the structure of the bacterial communities. Previous studies showing that silt- and clay-sized particles are associated with bacterial communities with different structure and greater diversity compared to sand-sized particles (Ranjard *et al.* 2000, Sessitsch *et al.* 2001, Torsvik and Ovreas, 2002) have not taken into account their different mineral type and composition, which is known to affect bacterial community structure (Carson *et al.* 2009). In this study, because soil texture was altered by adding silt- and clay-sized particles of quartz to a quartz based sand, it was possible to attribute the effect of texture on bacterial communities to the altered physical conditions in soil.

References

- Carson JK, Campbell L, Rooney D, Clipson N, Gleeson DB (2009) Minerals in soil select distinct bacterial communities in their microhabitats. *FEMS Microbiology Ecology* **67**, 381-388
- Curtis TP, Sloan WT, Scannell JW (2002) Estimating prokaryotic diversity and its limits. *Proceedings of the National Academy of Sciences of the United States of America* **99**, 10494-10499.
- Dens EJ, Van Impe JF (2001) On the need for another type of predictive model in structured foods. *International Journal of Food Microbiology* **64**, 247-260.
- Deschesne A, Or D, Smets BF (2008) Limited diffusive fluxes of substrate facilitate coexistence of two competing bacterial strains. *FEMS Microbiology Ecology* **64**, 1-8.
- Long T, Or D (2005) Aquatic habitats and diffusion constraints affecting microbial coexistence in unsaturated porous media. *Water Resources Research* **41**, W08408.
- Ranjard L, Poly F, Combrisson J, Richaume A, Gourbière F, Thioulouse J, Nazaret S (2000) Heterogeneous cell density and genetic structure of bacterial pools associated with various soil microenvironments as determined by enumeration and DNA fingerprinting approach (RISA). *Microbial Ecology* **39**, 263-272.
- Sessitsch A, Weilharter A, Gerzabek M, Kirchmann H, Kandeler E (2001) Microbial population structures in soil particle size fractions of a long-term fertilizer field experiment. *Applied and Environmental Microbiology* **67**, 4215-4224.
- Tiedje J, Cho J, Murray A, Treves D, Xia B, Zhou J (2001) Soil is teeming with life: New frontiers in soil science. In 'Sustainable Management of Soil Organic Matter' (Eds RM Rees, B Ball, C Campbell, CA Watson) pp. 393-412. (CAB International).
- Tilman DT (1982) (Resource Competition and Community Structure). (Princeton University Press: Princeton, USA).
- Torsvik V, Ovreas L (2002) Microbial diversity and function in soil: from genes to ecosystems. *Current Opinion in Microbiology* **53**, 240-245.
- Treves DS, Xia B, Zhou J, Tiedje JM (2003) A two-species test of the hypothesis that spatial isolation influences microbial diversity in soil. *Microbial Ecology* **45**, 20-28.

The effects of organic and conventional farming systems on selected soil properties of olive groves in central Greece

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Abstract

Olive tree cultivation is of great socioeconomic and environmental importance in the Mediterranean region. The maintenance of fertile soil is the basis on which a productive and sustainable agricultural system depends. In the present study a comparison between organic and conventional olive groves regarding the fertility of the soil in Magnesia Prefecture was conducted. In particular, soil samples were taken from two different farming systems (conventional and organic) of olive groves and analyzed for various soil parameters. A comparison between the mean values for soils under organic systems and their controls showed an overall improvement in the soil quality of the former, with increased contents of organic matter, available P, exchangeable K, NO₃, NH₄, CEC and pH values. Moreover, the organic matter, P and CEC were found to be correlated with the farming system and they are factors of distinguishing between the organic and conventional olive groves in the research area.

Key Words

Soil quality, olive groves, organic, conventional, Greece.

Introduction

Olive tree cultivation is of great social, economic and environmental importance in the Mediterranean region. Productive cultivated land is considered the land with good structure and drainage, having at the same time the ability to retain the suitable soil moisture and containing sufficient amounts of nutrients for the development of the plants (Stockdale *et al.* 2002). In assessing the fertility of a soil, not only its chemical properties but also its organic and physical characteristics should be taken into account, as these may be adversely affected by intensive soil cultivation practices over time (Werner 1997). This paper describes the characteristics of organic and conventional cultivated soils in an effort to investigate any changes in the soils of olive groves, resulted from applying the different farming practices. Here we present a preliminary analysis of selected soil properties.

Methods

The research was conducted in ten conventional and ten organic olive groves located in southwest Magnesia, Greece in 2009 (Figure 1). Olive trees planted in 1860 and were entered in organic management system in 1997. The soil texture was clay loam. The sampling was conducted in March by using a cylindrical sampler in depth of 0-30 cm. The number of samples taken was 10 for organic and 10 for conventional olive groves. Each sample consisted of 5 cores, well mixed on site, which were collected from different points in the field. The following characteristics of the soil were determined: soil texture (Bouyoukos 1951), organic matter (Nelson and Sommers 1982), ammonium nitrogen (Bremner 1960), nitrate nitrogen (Keeney and Nelson 1982), available phosphorus (Olsen and Sommers 1982), potassium (Thomas 1982), cation exchange capacity (Rhoades 1982b), pH (Mc Lean 1982) and CaCO₃ (Nelson and Sommers 1982). The fertilization applied in the area was that of calciferous nitric ammonia (26-0-0), 2 kg per tree, in the conventional olive groves and borax 200 gr per tree, potassium 1-1.5 kg per tree, and an amount of 50 kg per tree of digested manure originated from animals of organic farming were applied in the organic olive groves. Statistical analyses were performed using SPSS 18 Software. Mean comparisons were done using the one way ANOVA tests at $\alpha=0.05$ level. Some of data (CaCO₃, K, NO₃, NH₄) were log (x + 1) transformed prior to analysis to obtain normality. Correlation analysis [Pearson] and Regression analysis (Binary logistic) were used to test relationships between variables.

Results

The soil texture was very variable in the two farming systems as shown in Table 1. Consistent differences in the percentage of sandy /clay particles were not observed amongst the two farming systems even though the

organic farmed soils tend to be heavier textured (SCL vs CL). The difference in clay content was not significant ($F=3.11$, $p=0.09$, $df = 1$) between the two treatments.



Figure 1. The study area.

Table 1. Soil texture of olive groves. (*)

Mean	Organic	O1	O2	O3	O4	O5	O6	O7	O8	O9	O10	Mean
25/51	%Clay/Sand	17/56	21/44	32/43	29/46	39/26	24/59	24/67	24/45	20/61	22/65	25/51
CL	Type	SCL	CL	L	CL	CL	SL	SL	CL	SL	SL	CL
Mean	Conventional	C1	C2	C3	C4	C5	C6	C7	C8	C9	C10	Mean
26/56	%Clay/Sand	17/56	25/54	39/40	26/59	29/58	26/47	30/59	32/53	24/63	16/69	26/56
SCL	Type	SCL	SCL	L	SL	SL	SCL	SL	SCL	SL	SL	SCL

*Symbols: O: Organic olive groves, C: Conventional olive groves, SCL: sandy clay loam, CL: clay loam, L: loam, SL: sandy loam

Sand, silt and clay particles bound together by organic matter is a vital component of soil, influencing fertility, soil structure, workability and water holding capacity, as well as carbon storage. In the organic olive groves a significantly ($F=5.84$, $p = 0.02$, $df=1$) higher percentage of organic matter was observed (mean = 2.98, $SE=0.41$) in comparison to the conventional ones (mean = 1.77, $SE = 0.27$) (Figure 2). This may be due to fertilization with the manure of the animals and the cutting of weeds in the organic olive groves. Both of them favor the availability of organic matter in comparison to the conventional olive groves. Jiao *et al.* (2006) found that the annual addition of manure in amounts exceeding 30 t/ha, increased the organic C in the organic olive groves. No significant differences in soil pH were found between organic olive groves (mean = 7.01, $SE= 0.36$) and conventional ones (mean= 6.25, $SE= 0.27$) (Figure 3).

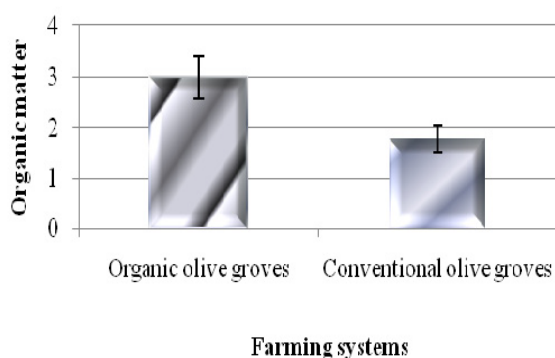


Figure 2. The effect of the farming system on the organic matter of the soil.

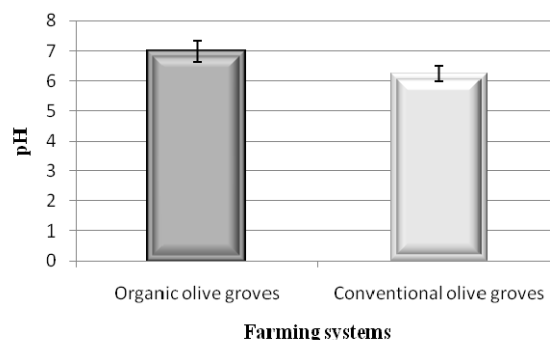


Figure 3. The effect of the farming system on the pH of the soil.

This may be due to the fact that the soil pH is mainly affected by the soil formation, the organic activity, the season of the year and the fertilization (Smith and Doran 1996). The mean concentration of K was higher in the organic olive groves (mean = 2.09, $SE = 0.12$) compared to the conventional ones (mean=1.98, $SE=0.06$). Regarding the mean concentration of K, no statistically significant difference was found ($F = 0.65$, $p = 0.42$, $df = 1$) between the two types of the olive groves. This may be due to the inorganic fertilization which was applied to the conventional olive groves and the organic fertilization which was applied to the organic ones (Schjonning *et al.* 2002). The mean concentration of P according to Olsen was significantly higher in the organic olive groves (mean = 3.95, $SE = 0.58$) than in the conventional ones (mean = 2.4, $SE =$

0.16) ($p < 0.05$) ($F = 6.53$, $p = 0.02$, $df = 1$) (Figure 4). This might be attributed to the fact that the manure application in organic sites mainly as a means of nitrogenous fertilization of the crop, may lead to the accumulation of P in the soil (Schwartz and Dao 2005), because the as the ratio N/P of the most types of manure is smaller than the corresponding ratio of the majority of the conventional crops (Whalen *et al.* 2001).

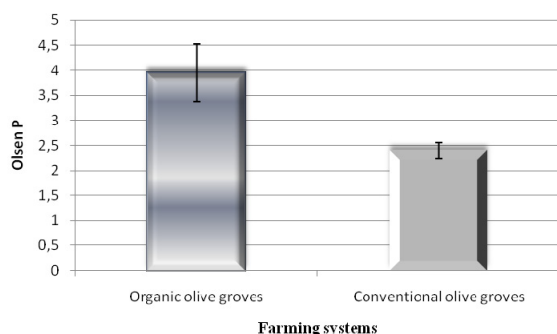


Figure 4. The effect of the farming system on the P according to Olsen.

The cation exchange capacity (CEC) of the organic olive groves was significantly higher (mean = 18.5, SE = 2.17) than the conventional ones (mean = 12.55, SE = 1.33), ($F = 5.42$, $p = 0.03$, $df = 1$). This result is probably due to the fact that in the organic agriculture the increased application of manure increases the cation exchange capacity of the topsoil layers (0-30cm) due to the increase in the organic matter (Eghball 2002). Soil pH is important for CEC because as pH increases the number of negative charges on the colloids increases, thereby increasing CEC. The mean concentration of NO_3 showed no statistically significant difference between the organic and the conventional olive groves, as it is shown by the level of significance in these two cases ($p > 0.05$) ($F = 3.01$, $p = 0.1$, $df = 1$). Similar results were obtained for NH_4 ($p > 0.05$) ($F = 0.55$, $p = 0.46$, $df = 1$). This parameter is related to a great extent to the concentrations of organic nitrogen and fungus biomass. Moreover, the mean concentration of CaCO_3 was not statistically significant different ($p > 0.05$) ($F = 1.03$, $p = 0.32$, $df = 1$) between the organic olive groves and the conventional ones. This result may be due to the soil texture in the two farming systems which are applied in the olive groves of the southwest Magnesia (Table 1). Correlations were found between organic and conventional sites in respect to organic matter and farming system (sig.=0.05), CEC and farming system (sig.=0.05), P and farming system (sig.=0.05), pH and CaCO_3 [$r = 0.85(**)$] (highly strong positive linear), sig.=0.00], P and NO_3 [$r = 0.86(**)$] (highly strong positive linear), sig.=0.04], CEC and organic matter [$r = 0.79(**)$] (strong positive linear), sig.=0.00], K and organic matter [$r = 0.71(**)$] (strong positive linear), sig.=0.00], organic matter and NO_3 [$r = 0.71(**)$] (strong positive linear), CEC and K [$r = 0.62(**)$] (waist positive linear), sig.=0.00], P and NH_4 [$r = 0.61(**)$] (waist positive linear), sig.=0.00], P and organic matter [$r = 0.55(*)$] (waist positive linear), sig.=0.00], NO_3 and NH_4 [$r = 0.57(**)$] (waist positive linear), sig.=0.00]. In the organic olive groves significant correlations were found K and organic matter [$r = 0.90(**)$] (highly strong positive linear), sig.=0.00], CEC and organic matter [$r = 0.86(**)$] (highly strong positive linear), sig.=0.00], NO_3 and P [$r = 0.86(**)$] (highly strong positive linear), sig.=0.00], pH and CaCO_3 [$r = 0.85(**)$] (highly strong positive linear), sig.=0.00], P and NH_4 [$r = 0.82(**)$] (highly strong positive linear), sig.=0.03], NO_3 and NH_4 [$r = 0.83(*)$] (highly strong positive linear), sig.=0.00], CEC and K [$r = 0.78(**)$] (strong positive linear), sig.=0.00], K and NO_3 [$r = 0.76(**)$] (strong positive linear), sig.=0.00], K and P [$r = 0.68(*)$] (waist positive linear), sig.=0.03], organic matter and NO_3 [$r = 0.68(*)$] (waist positive linear), sig.=0.03]. It will be of interest to follow these relationships as the transition period to organic practices proceeds. As regards the conventional olive groves the correlation was observed in CaCO_3 and pH [$r = 0.85(**)$] (highly strong positive linear), sig.=0.00], P and the organic matter [$r = 0.73(*)$] (strong positive linear), sig.=0.01], NH_4 and organic matter [$r = 0.71(*)$] (strong positive linear), sig.=0.02] of the soil.

Conclusions

Based on the data collected on soil parameters, it seems that the conversion to organic farming for the conventional olive groves is progressing satisfactorily in southwest Magnesia Central Greece, with a gradual improvement in soil quality as organic fertilizers are applied. The organic matter, P and CEC were found to be correlated with the farming system and they are factors of discrimination between organic and conventional olive groves in the research area. CaCO_3 , NO_3 , NH_4 , K and pH, although they showed higher mean values in the organic olive groves than in the conventional ones, the variance analysis showed that they are not factors of discrimination between the two farming systems.

References

- Bremner JM (1960) Determination of nitrogen in soil by Kjeldahl method. *Journal of agricultural science* **55**, 1-23.
- Eghball B (2002) Soil properties as influenced by phosphorus- and nitrogen based manure and compost applications. *Agronomy Journal* **94**, 128-135.
- Jiao Y, Whalen JK, Hendershot WH (2006) No-tillage and manure applications increase aggregation and improve nutrient retention in a sandyloam soil. *Geoderma* **134**, 24-33.
- Keeney D R, Nelson DW (1982) Nitrogen: Inorganic forms. In 'Methods of soil analysis. Chemical and microbiological properties'. (Eds AL Page, RM Miller, DR Keeney) pp. 167-179. (American Society of Agronomy Inc.: Madison, WI).
- Mc Lean EO (1982) Soil pH and lime requirement. In 'Methods of soil analysis. Chemical and microbiological properties'. (Eds AL Page, RM Miller, DR Keeney) pp. 199-224. (Eds AL Page, RM Miller, DR Keeney) pp. 167-179. (American Society of Agronomy Inc.: Madison, WI).
- Nelson DW, Sommers LE (1982) Total carbon, organic carbon and organic matter. In 'Methods of soil analysis. Chemical and microbiological properties'. (Eds AL Page, RM Miller, DR Keeney) pp. 539-580. (Eds AL Page, RM Miller, DR Keeney) pp. 167-179. (American Society of Agronomy Inc.: Madison, WI).
- Olsen SR, Sommers LE (1982) Phosphorus. In 'Methods of soil analysis'. (Eds AL Page, RM Miller, DR Keeney) pp. 403-430. (American Society of Agronomy Inc.: Madison, WI).
- Rhoades JD (1982b) Cation exchange capacity. In 'Methods of soil analysis. Chemical and microbiological properties'. (Eds AL Page, RM Miller, DR Keeney) pp. 149-158. (American Society of Agronomy Inc.: Madison, WI).
- Schjønning P, Christensen BT, Carstensen B (1994). Physical and chemical properties of a sandy loam receiving animal manure, mineral fertilizer or no fertilizer for 90 years. *European Journal of Soil Science* **45**, 257-268.
- Schwartz RC, Dao TH (2005) Phosphorus extractability of soils amended with stockpiled and composted cattle manure. *Journal of Environmental Quality* **34**, 970-978.
- Smith J, Doran JW (1996) Measurement and use of pH and electrical conductivity for soil quality analysis. In 'Methods for assessing soil quality'. (Eds JW Doran, AJ Jones) pp. 169-185. (American Society of Agronomy Inc.: Madison, WI).
- Stockdale EA, Shepherd MA, Fortune S, Cuttle S P (2002). Soil fertility in organic farming systems- fundamentally different. *Soil Use and Management* **18**, 301-308.
- Thomas GW (1982) Exchangeable cations. In 'Methods of soil analysis. Chemical and microbiological properties'. (Eds AL Page, RM Miller, DR Keeney) pp 159-165. (American Society of Agronomy Inc.: Madison, WI).
- Werner R, Matthew R (1977) Soil quality characteristics during conversion to organic orchard management. *Applied Soil Ecology* **5**, 151-167.