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Aggregation, carbohydrate, total and particulate organic carbon changes by cultivation of an arid soil in Central Iran

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Abstract
The objectives of this paper were to determine the response of soil quality indicators and organic carbon (OC) distributions within different aggregate classes to changes in land use from desert soils to cropland occurring in the Abarkooh plain, Central Iran. Composite soil samples of the desert soil, wheat and alfalfa fields were taken from three different depths and soil quality indicators, including aggregate stability (MWD), OC, carbohydrate, particulate organic carbon existent in macro (POC_{mac}) and microaggregates (POC_{mic}) in these soils were determined. Conversion of desert soils to croplands resulted in a significant decrease in electrical conductivity (EC), but an increase in MWD, OC, carbohydrate, POC_{mac} and POC_{mic} contents. OC content increased with increased aggregate size and the aggregate; OC ratio in cultivated and desert soils was highest in the 1-2 and <0.05 mm aggregates, respectively. The improvement soil quality in cropland is due to the long-term use of irrigation water and soil fertilization and also the poor SOM in desert soils. The results of this study further indicate the POC_{mac}, POC_{mic} and POC_{mac}/POC_{mic} ratio are sensitive parameters that reflect differences of soil aggregation, SOM quality and tillage intensity in soils of this region. These parameters are more reliable indicators of soil quality than OC in reclaimed desert soils.

Key Words
Soil quality, desert soils, cultivation, irrigation, soil aggregate, particulate organic carbon (POC).

Introduction
The desert soils in arid regions of Central Iran are characterized by low rainfall, low fertility, high evaporation and salinity. Also, soil water and salinity are crucial factors influencing crop production in these regions. Numerous field trials have demonstrated the effectiveness of leaching for salt removal. For example, Fullen *et al*. (1995) reported that use of irrigation water resulted in distinct and rapid improvements in the physical and chemical properties of reclaimed desert soils. Soil aggregate structure and stability are important factors that contribute to sustainable soil quality (Shepherd *et al.* 2002). Numerous studies have shown that conversion of native ecosystems to agriculture, especially in tropical and temperate regions have led to a negative impact on soil quality (Islam and Weil 2000). Nonetheless, information about the soil quality of desert soils and marginal cropland is scarce. Thus, the main objectives of this study were: (1) to analyze the effects of changes in land use from desert soils to croplands on soil quality indicators (2) to determine aggregate organic carbon (OC) ratio (OC in aggregate/OC in total aggregates) and also the distribution of OC in the aggregates of soils under desert and cropland.

Methods
Study area
The study area was located in Abarkooh plain at an elevation of around 1500 m above sea level, nearly 140 km southwest of Yazd, Iran (31° 18′ N, 53° 17′ E). The climate of the region is arid with a mean annual rainfall of 60 mm, potential evapotranspiration of 2800 mm and the temperatures ranging from 40 °C in summer down to -13 °C in winter. In the southeast Abarkooh plain, groundwater is mainly saline (1700–2500 μmho/cm), but could be exploited to meet crop water requirement. A flood-irrigation system was developed during 1979–1981 to reduce salinity levels in the root zone and increase water availability. Therefore, this research was conducted to understand the changes of soil quality and aggregation resulting only from cultivation of desert soils. For this purpose, three different land uses were chosen in the study area including virgin desert, alfalfa (*Medicago sativa* L.) in rotation with wheat, and wheat (*Triticum aestivum* L.) in rotation with fallow and barley. At the time of sampling, the vegetation of the desert soils was dominated by *Tamarix hispida*. In the cropland, mineral N (urea) and P (di-ammonium phosphate) fertilizers are usually applied for improving soil productivity.

Soil sampling and analysis
In June 2008, three blocks (each 60×60 m) were randomly selected from each of the two treatments (wheat and alfalfa fields) for soil sampling to produce nine pseudo replications. At each block, nine sub-samples at each depth of 0–10, 10–20 and 20–30 cm were taken and mixed as three composite samples. Adjacent to sampling areas of the two treatments, three uncultivated (desert soils) blocks were randomly selected and sampled in the same way. Overall, 81 composite soil samples were collected, comprising three land uses, three depths and nine replicates. After air drying, soil samples were sieved through 4 mm sieve size for aggregate fractionation and separation of POC, and the remaining was sieved through 2 mm sieve size for chemical analysis and particle size distribution. Soil electrical conductivity (EC) was measured in saturated extracts and OC with the Walkley & Black method, were determined. The content of dilute acid-hydrolysable carbohydrate (CH_{da}) in whole soils was determined by the phenol-sulphuric acid method of Dubois *et al.* (1956).

Fractionation of water stable aggregates and separation of POC_{mac} and POC_{mic}
The size distribution of soil aggregates was measured by wet sieving through a series of sieves (2, 1, 0.5, 0.25 and 0.05 mm). Also, the material passing the 0.05 mm sieve (<0.05 mm) was collected. For the separation of particulate organic matter (POM), aggregate fractions were combined into two groups: macroaggregate (0.25-2 mm) and microaggregate (0.05-0.25 mm). Soils from the bulked macroaggregate and microaggregate fractions were dried (50˚C) in the oven overnight and cooled in a decicator to room temperature. Then, 10 g of each aggregate fraction was dispersed in 30 ml sodium hexametaphosphate (5%) for 16 h on a reciprocating shaker at 120 recipricitions per minute. After dispersion, the suspensions were sieved through 0.05 mm sieve to separate sand particles and POM. The separation of POM by Loss on Ignition (LOI) was done following the procedure of Cambardella *et al.* (2001). The collected sand particles + POM were dried at 55˚C to constant weight, and then subjected to 450˚C for 4 h to measure POM by LOI method and POC estimated by multiplying the mass difference by 0.58.

Statistical analysis
The physical and chemical properties in the whole soils and aggregate fractions were repeated nine and three times, respectively. In each depth, one-way ANOVA was conducted to detect significant differences between land uses and, among the different aggregate size classes. Significant treatment means were separated using Duncan test at $p < 0.05$. Statistical procedures were carried out using the software package SPSS 15.0 for Windows.

Results and discussion

The electrical conductivity (EC)
The long-term use of irrigation water led to significant leaching of soluble salts from topsoil, therefore, soil EC was substantially lower in the alfalfa and wheat fields than in the desert soils. Desert soils contained the highest EC at all depths (Figure 1).

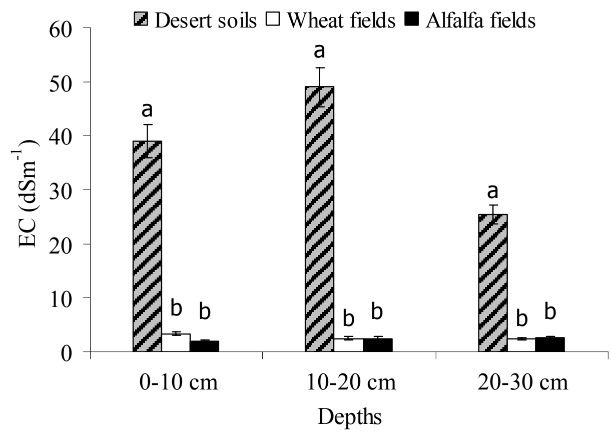

Figure 1. Electrical conductivity (EC) of soils under different land uses at the three depths. In a depth, similar letters indicate no significant difference among the land uses at p <0.05. Each value represents means ± S.E. (n=9).

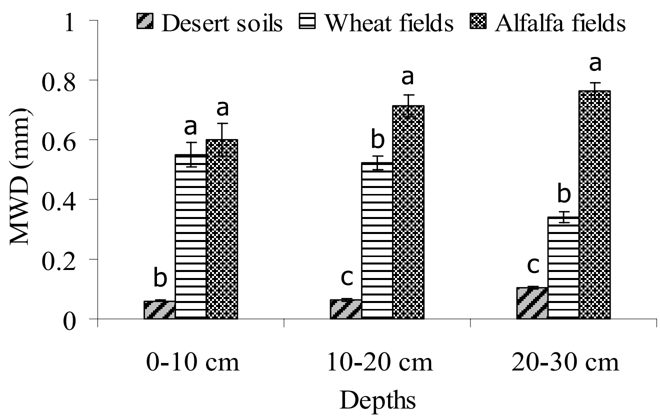

Figure 2. Mean aggregate stability (MWD) of soils under different land uses at the three depths. In a depth, similar letters indicate no significant difference among the land uses at p <0.05. Each value represents means ± S.E. (n=3).

The Water-stable aggregates distribution and stability

The most frequent aggregate fractions were the aggregates <0.05 mm for the desert soils, the microaggregates for wheat and the aggregates 1–2 mm for alfalfa at all depths (data not shown). Because of high soil EC in the desert soils, larger aggregates (>1 mm) are unstable and the formation of stable aggregates is very poor in these soils. Hence reclamation and cultivation of desert soils considerably increased the value of aggregate stability (Figure 2). The long-term use of irrigation water led to substantial leaching of soluble salts from topsoil and caused an improvement in the soil structure.

The OC and CH_{da} contents

The OC and CH_{da} contents were highest in the soil samples of alfalfa fields, intermediate in wheat fields and lowest under desert soils at all depths (Table 1). The lower OC and CH_{da} contents in desert soils may partly be attributed to reduced plant residues input to the soil because of limitation to plant growth in the harsh environment (low moisture and fertility and high salinity of soil). Salinity-induced degradation in arid soils is characterized by low soil OC values and indeed, the most saline soils had the lowest OC content (Yuan *et al.* 2007). Therefore the increases in OC and CH_{da} contents in cultivated fields were mainly related to higher carbon inputs from crop residues incorporated in the surface soil during improvement of soil property with irrigation. In agreement with our results, Fullen *et al*. (1995) found that the irrigation caused an increase in SOM content of reclaimed desert soils. On the other hand, soil fertilization with nitrogen fertilizer (urea) in cropland could increase the level of crop residue returned to the soil, leading to enhanced OC content. Similar findings were reported by Alvarez, (2005) who found that the effect of nitrogen fertilizer on SOM storage was very probably a consequence of the increase in residues returned to the soil where fertilizer nitrogen was used. Our results are in contrast with the findings of studies in tropical and temper regions that shows conversion of the virgin lands into continuous cultivation resulted in significant decrease in the content of SOM (Islam and Weil 2000). Indeed, these researches show that the most virgin lands (pasture or forest) in tropical or temperate regions had the high SOM content and cultivation resulted in significant reduction in inputs of plant residues. Nonetheless, our results revealed that in arid regions, virgin soils because of existing water limitation and high salinity had low productivity and generally are very poor in SOM. The irrigation and soil fertilization in cropland increases the level of crop residue returned to the soil.

Table 1. OC and CH_{da} contents of soils under different land use systems at the three depths.

Depths (cm)	OC (g/kg soil)			CH_{da} (g/kg soil)		
	Desert soils	Wheat fields	Alfalfa fields	Desert soils	Wheat fields	Alfalfa fields
0-10	0.56 C (0.03)	2.88 B (0.26)	3.80 A (0.30)	0.03 C (0.01)	0.78 B (0.07)	1.0 A (0.09)
10-20	0.72 C (0.06)	2.18 B (0.21)	3.30 A (0.27)	0.11 B (0.01)	0.60 A (0.06)	0.72 A (0.05)
20-30	0.26 C (0.02)	1.76 B (0.18)	3.01 A (0.27)	0.08 B (0.02)	0.47 A (0.06)	0.57 A (0.05)

OC; organic carbon, CH_{da}; dilute acid-hydrolysable carbohydrate, Means and S.E. (n= 9). In a row, similar letters indicate no significant difference between the land uses p <0.05.

The POC_{mac} and POC_{mic} contents

In desert soils, because the proportion of the macroaggregates for separation of POM was scant, the POC_{mac} in these soils was not determined. Also, the POC_{mic} contents in these soils were very low. Data on POC_{mac} and POC_{mic} contents (g/kg soil <4 mm) indicated that whereas the POC_{mac} contents were significantly higher in the alfalfa than in the wheat, POC_{mic} contents for alfalfa and wheat fields were not statistically different mainly due to large proportion of soil in microaggregate of soils under wheat (Table 2). In most of the cases, the POC_{mac} content was consistently higher than the POC_{mic}. These results are in agreement with the observation of John *et al.* (2005) who found that in cultivated soils, the organic matter content of macroaggregates is considerably greater than that of microaggregates. Since soil quality is dependent upon organic matter content, our results showed that land use changes from desert soils to cropland have resulted in positive influences on SOM components and these demonstrate that cultivation of desert soils that are very poor in SOM, could improve soil quality. At all depths, the alfalfa fields showed a higher POC_{mac}/POC_{mic} ratio compared to the wheat fields. This could be attributed to different annual organic matter input and the decomposing of crop residuals and SOM, due to annual tillage in wheat fields. The annual ploughing and disturbing the soil in wheat fields versus soil ploughing every 6-7 years in alfalfa fields may have led to the transfer of SOM from the macroaggregates to microaggregates. John *et al.* (2005) found that OC in macroaggregates is younger than OC in microaggregates; consequently POC_{mac} is younger and more labile than POC_{mic}. This suggested that the SOM in alfalfa fields is more labile than organic matter in wheat fields. It was concluded that POC_{mac}, POM_{mic} and POC_{mac}/POM_{mic} ratio are sensitive parameters that reflect differences of soil aggregation, organic matter quality and tillage intensity in soils of this region. These

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results showed that POC_{mac}, POC_{mic} and POC_{mac}/POM_{mic} ratio are more reliable indicators of soil quality than OC in reclaimed desert soils.

Table 2. POC_{mac} and POC_{mic} contents in cultivated soils at the three depths.

Depths (cm)	POC (g/kg soil <4 mm)			
	Alfalfa fields		Wheat fields	
	POC_{mac}	POC_{mic}	POC_{mac}	POC_{mic}
0-10	2.35±0.24a	0.36±0.03	0.57±0.06	
10-20	1.25±0.15a	0.26±0.02	0.65±0.05	0.27±0.03c
20-30	0.84±0.07a	0.25±0.03	0.4	

POC_{mac}; particulate organic carbon in macroaggregate, POC_{mic}; particulate organic carbon in microaggregate. Means ± S.E. (n=3). Similar letters in rows indicate no significant difference at p <0.05.

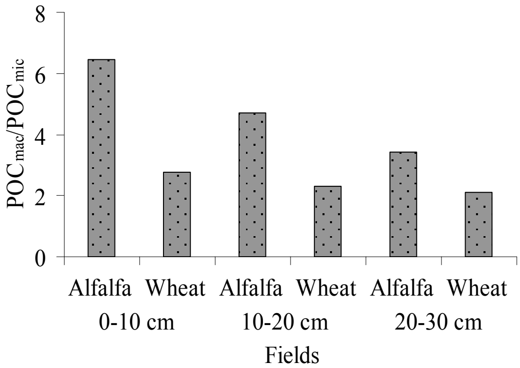

Figure 3. POC_{mac}/POM_{mic} ratios of cultivated soils at the three depths.

The OC content and aggregate OC ratio in aggregates

In the desert soils, OC content and the aggregate OC ratio (ratio of OC in aggregate to OC within total aggregates) were highest in the fraction <0.05 mm (data not shown). Conversely, in both cultivated soils, OC content was highest in aggregates 2-4 and 1-2 mm and the 1-2 mm aggregates had the highest aggregate OC ratio for the two cultivated soils. This indicated the importance of 1-2 mm aggregates, which stored more than 30-48% and 20-26% of the OC in soils under alfalfa and wheat fields, respectively. The general trend showed that OC content, in the cultivated soils, increases as aggregate size increased from 0.05 to 4 mm diameter (data not shown).

Conclusion

This study concluded that water deficit and soil salinity have larger roles in reducing soil quality than agricultural practices such as tillage. The results of the present study show the positive influences that land use conversion from desert soils to cropland might have on indicators of soil quality. The results of this study further indicate the POC_{mac}, POM_{mic} and POC_{mac}/POM_{mic} ratio are sensitive parameters that reflect differences of soil aggregation, SOM quality and tillage intensity in soils of this region. These parameters are more reliable indicators of soil quality than OC in reclaimed desert soils. In arid ecosystems, cultivation of desert soils that are very poor in SOM could improve soil quality.

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A micromorphological approach to soil rhizosphere characterisation of an agro-pastoral environment

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Abstract
This research aims at characterising some aggregates and their intra-porosity in paired pasture/maquis plots in a Mediterranean area (Sardinia, Italy). Surface horizons were sampled for thin sections analysis. Macroaggregate shape was compared and intra-porosity was studied for representative aggregates at different depths. The interaction of root activity and grazing under pasture and maquis cover is discussed.

Key Words
Microporosity, aggregate shape, root activity, deforestation, overgrazing, Mediterranean.

Introduction
Soil structure and in particular the pore space system has been the concern of numerous earlier studies dealing with the effect of cultivation techniques and crop types experimented at field sites. Recent long-term-experiments have documented the evolution of specific features related to land use, namely as the spheroidal or the onion skin, and vermicular microstructures (Kapur *et al.* 1997). Stoops' (2003) statement on the existence of infinite numbers of microstructure types at the same and at different scales of observation as well as the commonly occurring phenomenon of juxtaposition on the same level of observation, manifests the difficulty in determining the suite of microstructure typologies in any soil system. The present study aims at characterising some aggregates and their intraporosity in three couples of paired pasture/maquis plots in a Mediterranean area (Irgoli Municipality, Sardinia, Italy). Profiles studied in this paper were earlier described and soil degradation indicators were determined to assess the land use impact by Zucca *et al.* (2009). That study described the unsustainability of the current agropastoral practices, based on clearing of the natural vegetation and by periodic (every 3 to 5 years) ploughing and grazing.

Materials and methods
The thin sections of this study were collected from the same profiles of Zucca *et al.* (2010) and synthetically described in Table 1. Surface horizons were sampled for analysis and thin sections preparation. The soils were developed on "coarse-grained Palaeozoic granites associated with pegmatites and microgranites; white granites with micas/chlorites and their migmatites", generally termed as metamorphic schist or gneiss due to their uniform nature that developed following metamorphism. Soils studied are shallow (no more than 20-35 cm to the lithic contact), sandy loam, loam, or sandy clay loam, weakly acid moderately saturated. Pasture soils show different degrees of erosion and degradation. The micromorphological approach adopted for this study consisted of the determination of the shape (regularity/roundness increases to around 1.00) and the intra-porosity of selected aggregates, that may reflect the characteristics of the horizons and pore distribution in the rhizosphere (A and/or AB horizons). The description of the microstructure was undertaken according to the relative distribution of coarse to fine constituents as stated by Stoops and Jongerius (1975) later adapted by Bullock *et al.* (1985).The study was conducted on thin sections (8x5 cm) obtained from 6 undisturbed samples collected from the surface horizons of the different profiles. The thin sections were divided into 3 parts from 0 (surface of the thin section) up to 2 cm, from 2 to 4 cm and from 4 to 6 cm (Figure 1). In each part, representative aggregates (up to a size of 4.9 x 3.7 mm, corresponding to the maximum extent allowed by the optical acquisition system), were digitized for intra-porosity measurements. The analysis was undertaken according to Pagliai *et al*. (2004) and Pagliai (1988) by the software Image Pro Plus. The shape of the aggregates was determined by the Perimeter Convex/perimeter relationship.

Results and discussion
Profile 1: 2AP (Pasture)
The porphyric single space matrix and the dominant moderately developed incomplete subangular fine to

very fine weakly to moderately accordant microstructure (units/aggregates of 3 to 5 mm size) developing from the strongly weathering rock matrices with a continuous rock/soil phase and numerous sinuous bifurcating pores. Pores of <50 micron size increased at 0-2 cm and 2-4 cm depths, whereas a decrease of this size and an increase of larger pores were determined at 4-6 cm. This may reflect the higher influence of the roots at the top 4 cm and a probable compaction at the lowermost part of the surface horizon. The higher amounts of the elongated pores in this size range may also relate to the root action in the rhizosphere. The sum of the intra-aggregate porosity also indicates to the increased overall porosity in the 0 to 2 cm depth. This ties in well with the shape of the aggregates developing more rounded/regular in the 0 to 4 cm depths compared to the irregular aggregates of the 4 to 6 cm (Figures 2, 3a and 3b).

Table 1. The studied soil profiles. P = Pasture; Md = degraded Maquis. Classification according to USDA (1999).

	Plot	Profile N	Horizons	A hor. depth	Classification	% C	C/N
Site 2A; pastures cleared in late 1980s; slope 40%	P	1	AC	11cm	Lithic Xerorthents	2.0	25
	Md	2	ABwBC	10cm	Lithic Dystroxerepts	6.8	16
Site 1A; pastures cleared in 1970s; slope 35%	P	4	AC	9cm	Lithic Xerorthents	2.6	14
	Md	5	AC	10cm	Lithic Xerorthents	5.3	22
Site 1B; pastures cleared in 1970s; slope 30%	P	6	ApC	18cm	Lithic Xerorthents	2.6	15
	Md	7	AC	10cm	Lithic Xerorthents	5.4	20

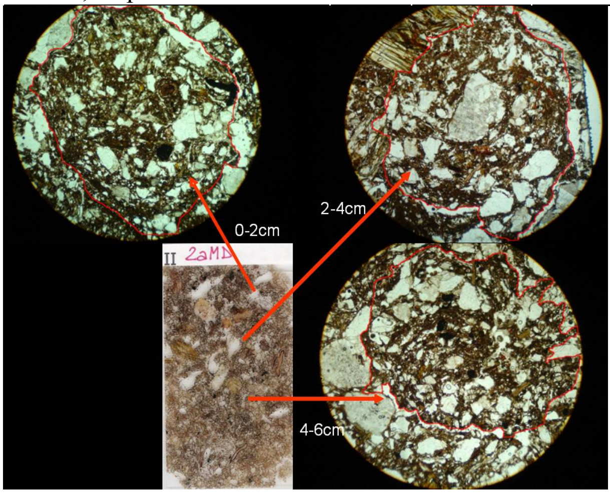

Figure 1. Selection of representative aggregates for the intra-porosity analysis of profile 2 (2AMd).

Profile 2: 2AMd (Degraded maquis)

The porphyric double space to enaulic matrix of crumby microstructure with voids and channels consists of moderately developed aggregates of 0.1 to 3 mm sizes. Pores of <50 micron size are dominant throughout the 0-6 cm part of the rhizosphere, followed with an increase of 100 to 200 micron pores in the 2 to 4 cm depth. The lowermost 4 to 6 cm part contains all sizes of pores with the dominant 50 to 200 micron size range (Figures. 1 and 3a and 3b). The elongated pores are higher in the 0 to 4 cm range, whereas the irregular (angular) increase at 4 to 6 cm. The situation for the pore shape distribution in this horizon reflects the effect of the root activity down to 4 cm. The gradual increase of the overall porosity may also indicate a strong root activity increase with depth. Aggregate shape tends to remain more rounded-regular at the 0-4 cm depth and shifts to higher irregularity in 4 to 6 cm depth (Figure 2).

Profile 4: 1AP (Pasture)

The porphyric single space matrix and microstructure is characterised by channels with moderately developed weakly separated aggregates of 0.1 to 3 mm sizes revealing a slight trend of decrease of regularity (roundness) with depth. Image analysis revealed that pores <50 micron were dominant throughout the 0-6 cm depth of the rhizosphere (as it is in the profiles above), with decreasing number of irregular pores from 0 to 6 cm, and an increase of elongated (micro-cracks) in 2 to 4 cm. The regular, irregular and elongated pores were about at equal amounts at 4 to 6 cm depth. The overall porosity of this horizon seems to be the lowest among all the studied horizons as is also reflected by the virtual absence of the majority of pore sizes (Figure 3a). The aggregates also show a slight decrease of regularity (roundness) with depth. The effect of the roots seems to be higher in mid horizon as manifested by the increasing elongated pores (Figures. 2 and 3b).

Profile 5: 1AMd (Degraded maquis)
The porphyric single space matrix contains a crumby subangular blocky to irregular microstructure and moderately distributed voids and canals. The moderate to rare distribution of the voids is also reflected by the sum of the low intra-aggregate porosity compared to 2AP and 2AMd (Figure 3a). The strongly developed and weakly separated aggregate sizes are 0.1 to 3 mm in size and intergrade to irregular with depth (Figure 2). The dominant pore size is <50 microns throughout the thin section with the elongated ones intergrading to irregular, similar to the aggregates, with depth (Figure 3a and 3b).

Profile 6: 1BP (Pasture)
Porphyric single space to enaulic matrix and microstructure with intergranular microaggregates intergrade to strongly developed, weakly separated crumby (0.05 - 3 mm size). Pores of <50 micron size (dominantly elongated) were determined to be dominant from 0 to 4 cm, whereas the >500 micron irregular pores, expected to increase permeability, increased at 4 to 6 cm depth. The shapes of the aggregates intergrade to irregular, parallel to the pores, from more rounded with depth at 4 to 6 cm (Figures 3a, 3b and 2).

Profile 7: 1BMd (Degraded maquis)
The enaulic to porphyric single space matrix with frequent crumby intergranular microstructure and strongly developed, weakly separated microaggregates of 0.05 - 2 mm size manifest an increased regularity with depth in contrast to the other horizons (Figure 2). The dominantly elongated <50 micron pores (0 to 2 cm) and almost all the other sizes of pores show a decrease at 2 to 4 cm and a consecutive increase at 4 to 6 cm depth as also reflected in the overall porosity (Figures 3a, 3b and 2).

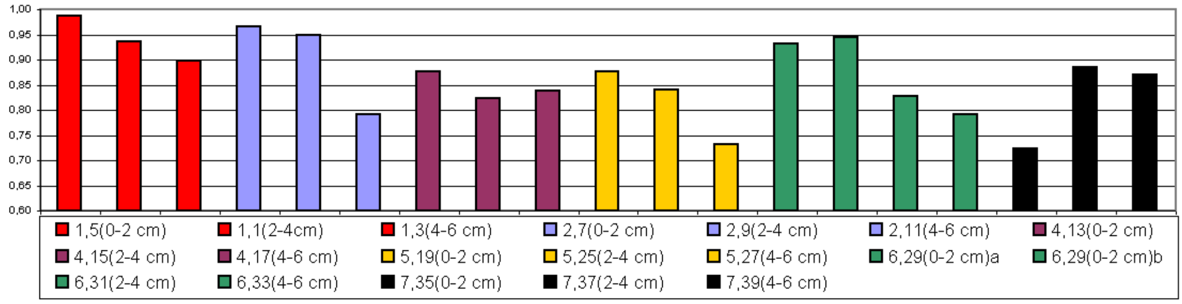

Figure 2. Shape analysis of the aggregates (regularity/roundness increases towards 1.00 per Convex/Perimeter).

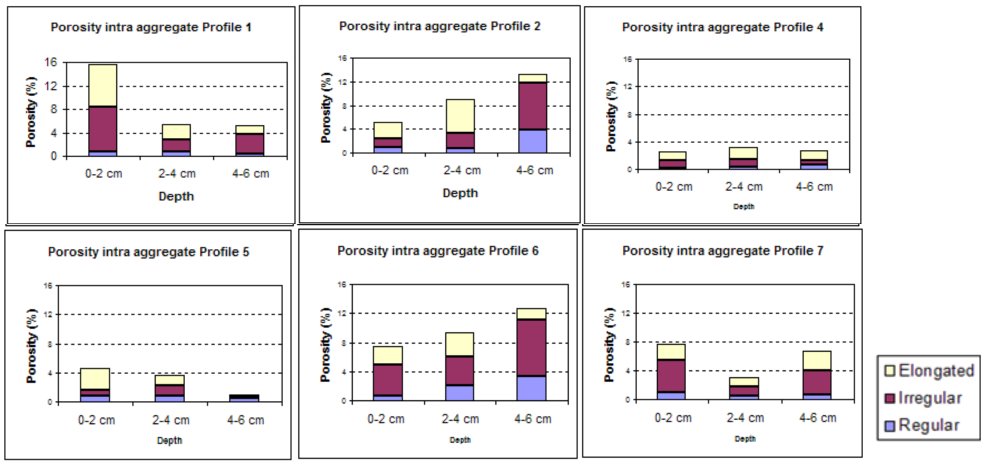

Figure 3a. The sum of the total porosity.

Conclusions
The sum of the total porosity of 2AMd (Maquis) and 1BP (Pasture) revealed an increase in pore percentage from 0 to 2 cm to 4 to 6 cm depth reflecting the increasing activity of the roots with depth, whereas 2AP (Pasture) and 1AMd (Maquis) show an opposite trend, which may point out to the decreasing influence of the roots and the higher effect of the finer 0-2 cm depth roots. The fluctuations in total porosity, also in the light of the 1AP and 2BMd values, seem to be related both to vegetation cover type and to the effects of grazing, which might induce or inhibit the development of the under and/or over-ground biomass. The <50 micron pores are dominant in 1AP (all layers), 2AMd (all layers), 2AP (0-2, 2-4 cm), 1AMd (all layers), 1BP (0-2, 2-4 cm) and 1BMd (all layers), whereas in 2AMd and 2AP, the 100 to 200 micron pores are high at 2-4 cm and 0-2 cm depths. The pores of 400-500 and >500 are dominant in 1BP, where the latter (4-6 cm) is the

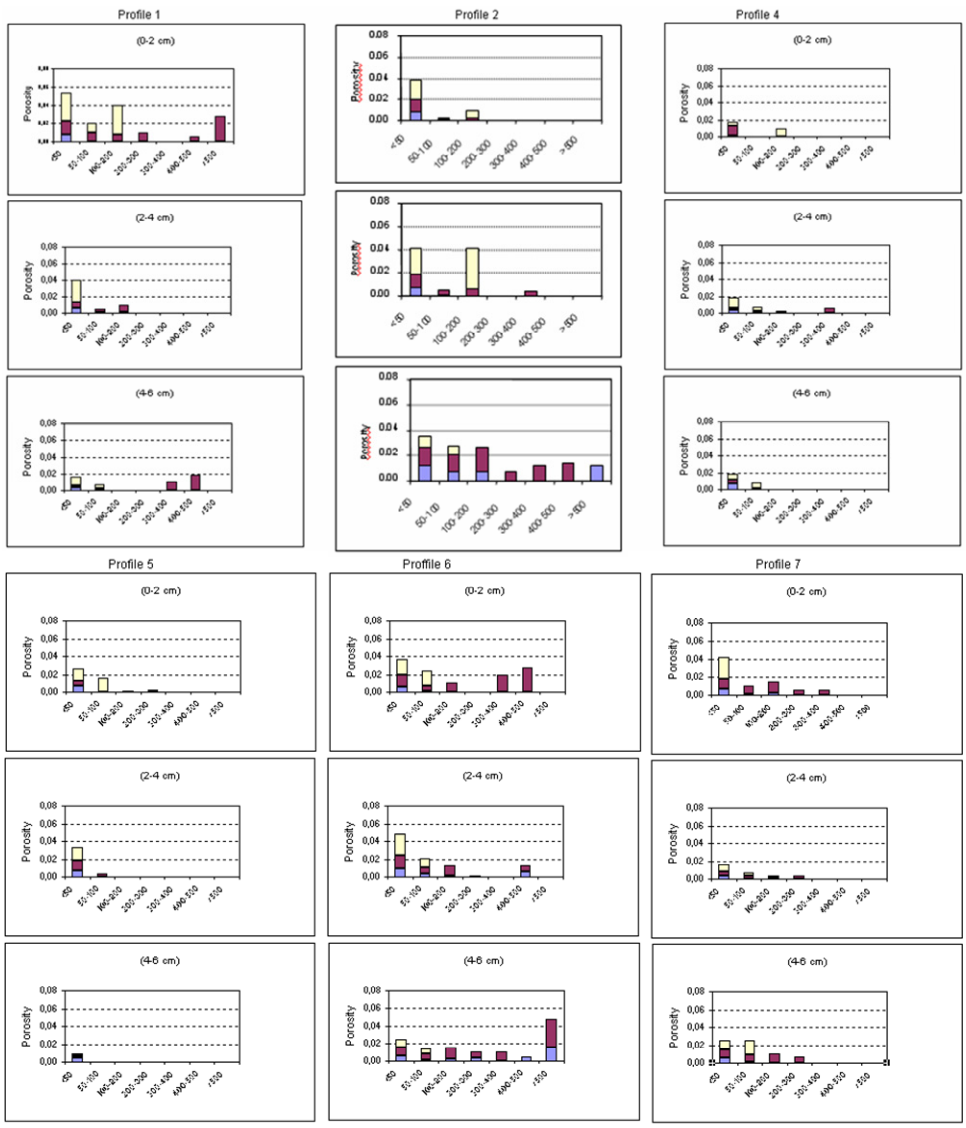

Figure 3b. The pore characteristics of the soils, by pore size class (in microns).

highest compared to the other profiles. The dominance of pores <50 micron under pasture and the highest values in the 0-2 and 2-4 depths of 2AP and 1BP, can indicate a positive contribution of the pasture vegetation to soil porosity but also a relative surface compaction effect due to grazing. The shape analysis of the aggregates of 2AMd, 2AP and 1BP also manifests the importance of pasture vegetation versus maquis.

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Biological properties of paleosols and present-day soils in Arkaim and its surrounding area, south Urals, Russia

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Abstract
We investigated biological properties of paleosols in comparison to those of present-day soils in Arkaim, Kizilskoe and Sintachta, south Urals, Russia. Soil microbial biomass C and N were detected not only in present-day soils but also in paleosols. The past cultivation history such as plowing and differences in parent materials (e.g., illite-montmorillonitic vs. kaolinitic) had significant effects on microbial biomass and activities of microbes. These results indicated that effects of human activity in the past (20 years ago) on soil microbes still remained and varied depending on the type of parent material. Soils originated from a riparian topography emitted more CO_2 and CH_4 as compared with extremely dried soils in grassland landscapes in Arkaim. N_2O flux was positive (1.3 mg N m^2 /h) on average in a soil with middle water content (28%), while almost zero in soils with minimum (11%) and maximum water content (33%).

Key Words
Greenhouse gasses, land-use change, microbial biomass, paleosol.

Introduction
On the brink of the third and second millennia B.C., a strong center of cultural genesis formed in the South Urals, Russia, with the powerful copper-ore base of the Trans-Urals plain that brought achievements in the field of metallurgy and consequent rapid economic and military developments in this area. The places of Arkaim, Kizilskoe and Sintachta, belonging to this area, are the examples of typical ancient fortresses developed in the period. We conducted a field investigation and sampling of paleosols and present-day soils in Arkaim, Kizilskoe and Sintachta, as a part of research collaboration between JSPS and RFBR Joint Research program. In this paper, we report biological properties of paleosols and present-day soils in relation to greenhouse gaseous emissions as affected by human activities, such as plowing, parent materials, and soil water content.

Materials and methods
Study area
The studied area is located in the Eurasian steppe zone. Fifteen soils located in Arkaim, Kizilsokoe and Sintachta in the administrative region of Chelyabinsk in south Urals, Russia (52^o N, 58 -60^o E) were investigated. Soils were sampled in August 2009, by taking samples from the upper two horizons: surface and sub-surface. Paleosols in this region were buried under the ancient embankment and artificial mounds. Each paleosol was sampled after removing upper present-day soil. The distance between paleosol and present day soil of each site was within 1 km. Soils plowed 20 years ago were sampled as plowed soil in Arkaim. Distances between plowed and unplowed soils were within 1 km. Soils with different water contents were sampled in riparian grassland near Arkaim.

Chemical and microbial biomass analysis
Fresh soil samples were sieved (2 mm) and analysed for soil soluble carbon (C) and nitrogen (N) and microbial biomass C and N within no longer than 1 month. Soil microbial biomass C and N were determined by chloroform-fumigation extraction method (Vance *et al.* 1987). Fresh soils were pre-incubated aerobically for 10 days and fumigated with alcohol-free chloroform at 25^{o}C for 24h. Then fumigated and non-fumigated soils were extracted with 0.5M K_2SO_4, and soluble organic C and soluble N concentrations were determined using a TOC analyser (TC5000, Shimadzu, Kyoto, Japan) and the peroxide-disulfate digestion/colorimetric method (Hayashi *et al.* 1997). Soil water content was determined by the oven drying method (Foster 1995).

DNA analysis
Total DNA from the samples was extracted using the FastDNA SPIN for Soil Kit (Qbiogene, California, USA) according to the manufacturer's protocol, and then extracted DNA was stored at -20 ^{o}C. *nirK* gene and 16S rRNA gene of methanogenic archaea in soils with different water contents were amplified by the PCR method using with a thermal cycler (Dice TP600, Takara bio Inc., Shiga, Japan) and suitable primer for each gene and DNA polymerase (Ex taq DNA polymerase Hot start version, Takara Bio Inc.).

Gas analysis
Carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) fluxes in the field were determined by the closed chamber method (Inubushi *et al.* 2003). The concentrations of CO_2, CH_4, and N_2O in sampled vials were analysed using gas chromatographs (GC 14B, Shimadzu) equipped with a thermal conductivity, electrons capture, and a flame ionization detectors, respectively. Each gas flux was calculated from the change in each gas flux concentrations over time.

Results and discussion
Chemical and microbial characteristic in paleosol and present-day soil
In paleosol and present-day soil of Arkaim, Kizilskoe and Sintachta, soil water contents were in the range of 1.6-5.8%. Soil microbial biomass was detected not only in present-day soil but also in paleosol, with larger values in present-day soil than in paleosol (Figure 1). Microbiological investigation of 100 steppe paleosols of Russia showed that in paleosols microbial communities preserved their existed communities from the time of archaeological monument construction (Blagodatskaya *et al.* 2003; Khomutova *et al.* 2004). Inubushi *et al*. (2005) discussed existence of microbial biomass in the buried humic horizon (i.e. ancient surface layer) of Andisol in Japan. They investigated vertical distribution of microbial biomass in Andisol profile and found higher amount of microbial biomass in deeper soil layers near the buried humic horizon. They reasoned that significant amount of biologically available organic matter remaining under semi-anaerobic conditions, which reversed to aerobic conditions during pre-incubation, or due to substrates leaching from surface layers and accumulating in deeper layers. In Arkaim and surrounding area, however, later reason is unreasonable because evaporation (450-650 mm /year) excedes precipitation (300-360 mm /year). Therefore, microbial biomass in paleosol in this study may be due to aerobic reverse of biologically available organic matter remaining in semi-anaerobic conditions during pre-incubation.

Effect of human activity on soil chemical and microbial characteristics
In soils of illite-montmorillonitic and kaolinitic parent material, soil water contents were in the range of 4.0-7.4%. Soluble C and N in surface of forest soil of illite-montmorillonitic parent material were the highest (336 and 63 mg/kg dry soil, respectivly). Microbial biomass of unplowed soils tended to be larger than of plowed soils (Figure 2). Figure 3 shows various gaseous fluxes as indicators of microbial activities in plowed or unplowed soils of illite-montmorillonitic or kaolinitic parent material. CO_2 was emitted in all types of soil. In soils of illite-montmorillonitic parent material, CO_2 emission in forest soil was larger than that emission in plowed and unplowed grass soils. In soils of kaolinitic parent material, however, there were no significant difference between CO_2 emissions in plowed and unplowed soil (p>0.05, t-test). CH_4 was taken in most soils. Soils of unplowed forest and grass on illite-montmorillonite parent material took in larger amount of CH_4 than plowed grass on the same parent material. However on kaolinitic parent material, the plowed soils took in the larger amount of CH_4 contrast with unplowed. Positive N_2O flux was only detected in plowed grass of illite-montmorillonitic parent material. Additionally, in soils of kaolinitic parent material, taken N_2O by plowed soil was larger than unplowed soil. These differences of microbial biomass and activities between plowed and unplowed soils indicated that effects of human activity (e.g., the plow more than 20 years ago) on soil microbes still have been remaining. Additionally, results of differences of microbial condition affected by human activity between illite-montmorillonitic and kaolinitic soils indicated that effects of human activity on soil microbes varied with the type of parent material.

Microbial characteristics of soils with different water content
In soils with different water content, CO_2 and CH_4 emissions were larger in soil with more water content. N_2O flux was positive in average only in a soil with middle water content (28%), while almost zero in soils with minimum (11%) and maximum (33%) water content. Total DNA from soils with different water content were extracted and amplified by PCR method using with primers for *nirK* and 16S rRNA gene of methanogenic archaea. DGGE analysis is undergoing.

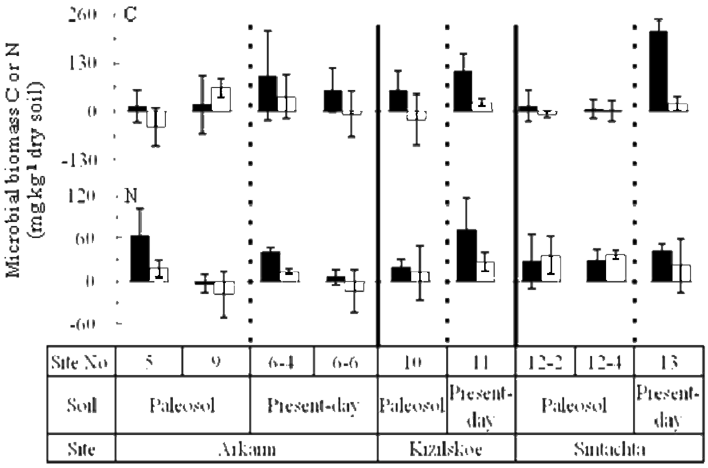


Figure 1. Microbial biomass C and N in surface (■) and subsurface (□) layer of paleosol or present-day soil of Arkaim, Kizilskoe or Sintachta.

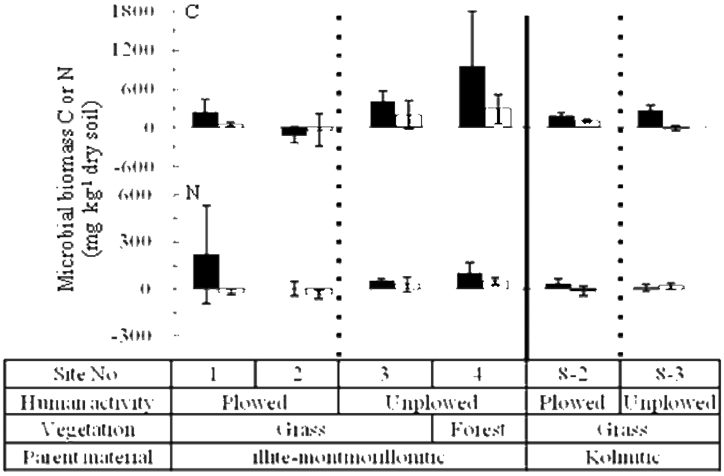


Figure 2. Microbial biomass C and N in surface (■) or subsurface layer (□) of plowed or unplowed soil of illite-montmorillonitic or kaolinitic parent material of Arkaim.

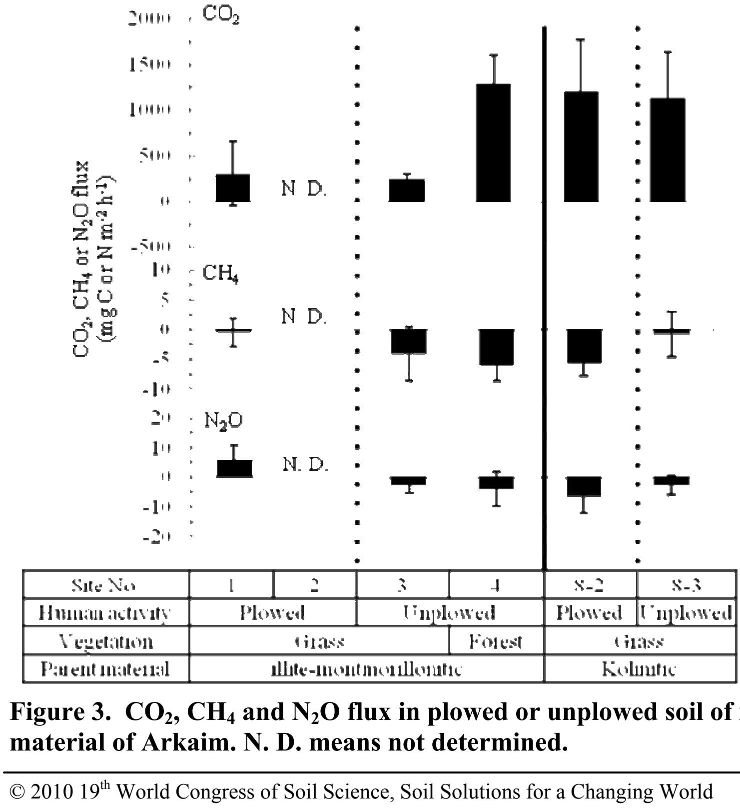
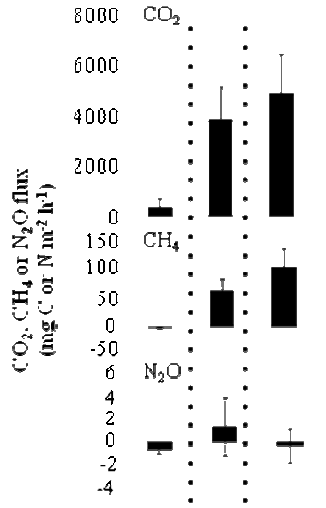


Figure 3. CO_2, CH_4 and N_2O flux in plowed or unplowed soil of illite-montmorillonitic or kaolinitic parent material of Arkaim. N. D. means not determined.



Site No	14-1	14-2	14-3
Water content (%)	11	28	33

Figure 4. CO_2, CH_4 and N_2O flux in soils with different water content of Arkaim.

Conclusions

The presence of microbial biomass in the paleosol horizon of Arkaim, Kizilskoe and Sintachta may be due to the aerobic reverse of biologically available organic matter remaining in semi-anaerobic conditions during pre-incubation. In soils of illite-montmorillonitic and kaolinitic parent material, differences of microbial biomass and activities between plowed and unplowed soils indicated that effects of human activity (e.g., the plow more than 20 years ago) on soil microbes remain. Additionally, results of differences of microbial condition affected by human activity between illite-montmorillonitic and kaolinitic soils indicate that effects of human activity on soil microbes varied with the type of parent material. In soils with different water content, CO_2 and CH_4 emissions were larger in soil with more water. N_2O flux was layer (1.3 mg N m^2 /h) on average in a soil with middle water content (28%), while it was almost zero in soils with minimum (11%) and maximum (33%) water contents.

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Carbon dioxide and nitrous oxide emissions associated with tropical peatland degradation

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Abstract
Long term monitoring of CO_2 emission from the peat of natural forest (NF), fire-affected re-growing forest (RF) and cropland (KV) in Plangka Raya, Central Kalimantan, Indonesia has been conducted since 2002. The results showed that the average annual CO_2 emission was 14.8 tC/ha/y in NF, 9.2 tC/ha/y in RF and 25.5 tC/ha/y in KV. The tendency of the CO_2 emission agreed with that of annual peat subsidence. Based on the data of the proportion of peat decomposition and root respiration in CO_2 emission from peat, the value of peat decomposition per unit peat subsidence was estimated to be 4.3 tC/ha/cm in NF, 6.21 tC/ha/cm in RF and 5.91 tC/ha/cm in KV, and peat decomposition to peat subsidence accounted for 46 % in NF, 57 % in RF and 57 % in KV. These results suggest that peatland degradation accelerates peat decomposition. N_2O emission exponentially increased with an increase of peat decomposition. The result of regression analysis using the data of CO_2 emission in tropical peatland compiled by Hooijer *et al*. (2006) together with the data obtained in this study showed that peat decomposition was significantly increased by a fall of the ground water table. The slope of the regression equation for cropland was higher than that for other ecosystems (natural forest, re-growing forest and plantation).

Key Words
CO_2, N_2O, peat decomposition, peat subsidence, ground water table.

Introduction
Indonesia is the country where more than 80% of tropical peat occurs (27Mha). Since 1985, tropical peat swamp forests in the world have been cut for timber production and destroyed widely, along with frequent forest fire. Even now, peat swamp forest is undergoing deforestation at 1.5%/y, to lose 120,000 km^2 (45%) due to clear cut and drainage (Hooijer *et al.* 2006). Among the deforested areas, reclaimed land used for agricultural production is only 30%, and the others are abandoned areas turned into degraded and deforested lands. Tropical peat is composed of woody materials, therefore the peat soils have a relatively high permeability compared with mineral soil or boreal sphagnum peat. Due to this high permeability, a deforested tropical peat dome quickly loses subsurface water from the soil into drainage. The peat soil which loses subsurface water starts to dry, does not allow vegetative recovery and CO_2 emission derived from peat decomposition remarkably increases.
An estimated, total 554 MtC/y CO_2 is released from tropical peatland throughout South East Asia, of which 172 MtC/y is caused by drainage and peat decomposition and 382 MtC/y by burning during peat fire (Hooijer *et al.* 2006). This total amount of CO_2 from peat soil is more than annual CO_2 emission in Japan. It is necessary to note that estimated total CO_2 emission from peat soils due to peat decomposition is based on correlation between ground water table and CO_2 emission, which was created by using 41 data set described in 16 publications. As it is hypothesized that one particular water table is determined by land use across the board, the CO_2 emission estimated in this manner remains uncertaint and unreliable.
In tropical peatland, peat decomposition is one of the factors for peat subsidence. Therefore, measurement of peat subsidence is crucial for predicting CO_2 emission derived from peat decomposition. Peat subsidence occurs due to not only due to peat decomposition but also to shrinkage, compaction and consolidation. Stephens and Stewart (1976) demonstrated that subsidence rate measured in a peat swamp in Florida, USA, increased along with groundwater table depth and soil temperature. Murayama and Bakar (1996) reported that annual peat subsidence in an oil palm plantation area in Peninsula Malaysia ranged from 1.55 to 1.64 cm, of which 52 to 66% was estimated to be due to peat decomposition using 10 tC/ha of annual average CO_2 emission. Wösten *et al*. (1997) estimated that annual CO_2 emission from a cropland in the peatland in Peninsula Malaysia was 27 tC/ha/y, along 2 cm/y peat subsidence, based on the estimation of 60% contribution of peat decomposition to the peat subsidence and 0.1g cm^{-3} bulk density of the peat soil.

However, they also estimated that bulk density of peat soil varied in the range from 0.05 to 0.15 g cm^{-3}. Hence they further estimated that CO_2 emission varied in the range from 13 to 40 tC/ha/y. CO_2 emission from peat land is strongly related to subsidence, ground water table depth, temperature and bulk density of peat. Therefore, a study about the controlling factors of CO_2 emission is required to understand the condition of peatland degradation.

Through the progression of peat decomposition, nitrogen mineralization in peat soil also occurs. This resulting ammonia undergoes nitrification under exposure to oxygen and successive denitrification under anaerobic conditions, leading to N_2O emission. It has been reported that 239 kgN/ha/y of N_2O is emitted from peat soil of arable land in Central Kalimantan, Indonesia (Takakai *et al.* 2006).

Purposes of this study are to estimate relationships between CO_2 emission derived from peat decompositions and subsidence in peatland and to clarify N_2O emission associated with peat decomposition.

Materials and methods

Study sites

Long term monitoring for subsidence and CO_2 and N_2O emissions have been conducted since 2001 at three sites in the peatland of Kalampangan zone near Palangka Raya, Central Kalimantan Indonesia（2˚S, 114˚E）: .1) natural forest influenced by drainage (FT), 2) re-growing forest after forest fire in 1998 and 2002 (RF), and 3) cropland (cultivated since late 1970's) (KV). Peatland of Kalampangan zone is composed of inland dome peat formed in a zone of about 20 km width between Sebangau River and Kahayan River. Around 30 years ago, exploitation started in the zone. After 1996, large scale exploitation proceeded with rice paddy development project under previous Suhatuto administration. However, in 1997/1998 and 2002, large scale peat fires occurred.

CO_2 and N_2O emission

CO_2 and N_2O fluxes from the soil surface have been measured by using a closed-chamber method since 2002. An open bottom cylindrical stainless-steel chamber (18.5-21.0cm in diameter and 25cm in height）was used for the measurement. One day before the measurement, any green vegetation on the measurement plots was cut at ground level and removed. In this case, CO_2 flux consists of CO_2 derived from root respiration and soil organic matter decomposition. In order to measure CO_2 flux derived from only peat decomposition, a root excluded plot of 1m×1m was established. Root-proofing permeable sheet (BKS9812, TOYOBO, Osaka, Japan) was vertically inserted to a depth of more than 30 cm below the ground surface to inhibit re-growth of roots. Flux measurement was conducted at three replicates in cropland and six replicates at other sites. Measurement of CO_2 and N_2O fluxes were conducted according to Takakai *et al*. (2007).

Air temperature at the height of 1m was measured at the same time with gas flux measurements using a thermistor thermometer (CT220, CUSTOM, Tokyo, Japan) for the gas flux calculation.

Subsidence and water table

The water table was measured at the time of gas flux measurement at each site by using a perforated PVC pipe (80 mm diameter and two meter long) with holes of 4 mm diameter opened at the four sides of the pipe every 5 cm interval along the pipe. The PVC pipe was also used for measuring the subsidence. From 2003 in KV, an iron pipe (40 mm diameter) which was vertically inserted from the peat surface to the top of the buried mineral soil layer was used. Subsidence was obtained from the change in the height from the peat surface to top of the pipe measured several times in a year.

Peat soil characteristics

Peat soil samples were taken at every 10 cm interval in each site by using a semi-cylindrical peat sampler (47 mm diameter, 50 cm long). At that time, peat depth was measured by the sampler. Bulk density, total carbon content, organic matter content and ash content were measured at the laboratory.

Results and discussion

CO_2 emission

Cumulative CO_2 emission at each site from April, 2002 to November, 2008 is shown in Figure 1. KV showed largest cumulative CO_2 emission was shown as 164 tC/ha in KV, followed by 97 tC/ha in FT and smallest CO_2 emission of 58 tC/ha in RF. Annual mean CO_2 emission (±SD) was significantly different among the sites which was 14.8 (± 2.8) tC/ha/y in FT, 9.2 (± 2.9) tC/ha/y in RF, and 25.5 (± 7.6) tC/ha/y in KV. The sites with high CO_2 emission showed large subsidence. Annual mean subsidence rate was largest in KV (4.1 cm/y), followed by FT (2.2 cm/y) and smallest in RF (0.9 cm/y).

However, the CO_2 emission from soil includes root respiration and peat decomposition. But, in crop fields, there was no significant difference of CO_2 emissions between root intact and root excluded plots. This indicates that root respiration can be ignored. However, Melling (2005) indicated that there was significant difference of CO_2 emissions between root intact and root excluded plots in a forest, oil palm plantation and sago plantation. The proportion of CO_2 emission from root excluded plot to root intact plot was 66% on average (66 % in forest, 60 % in oil palm plantation and 72 % in sago palm plantation). Using the contribution of peat decomposition to total CO_2 emission (100% in croplands and 66% in other lands), peat decomposition was estimated to be 9.76 tC/ha/y in FT, 6.07 tC/ha/y in RF and 24.0 tC/ha/y in KV (Table 1). The ratio of peat decomposition to peat subsidence was 4.40 tC/ha/cm in FT, 6.77 tC/ha/cm in RF and 6.27 tC/ha/cm in KV. Using the bulk density of top layer and peat C content, the contribution of peat decomposition to peat subsidence (decomposability) was estimated to be 42 % in NF, 62 % in RF and 61 % in KV. This indicates that agriculture and peat fire make the peat more decomposable.
Hooijer *et al*. (2006) indicated a positive tendency that the

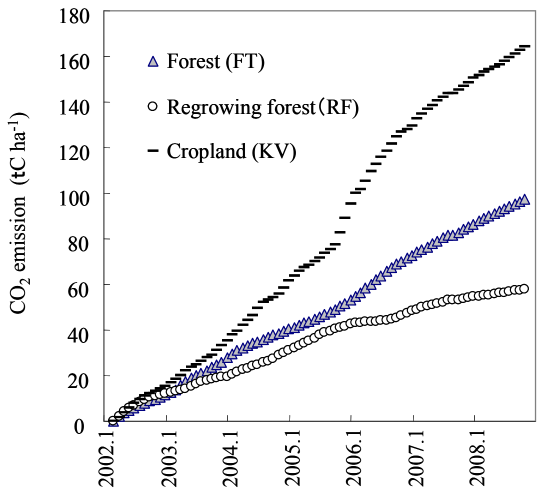

Figure 1. Cumulative CO_2 emission in Central Kalimantan, Indonesia after April 2002.

Table 1. Characteristics of CO_2 emission derived from peat decomposition in Central Kalimantan, Indonesia.

	Decomposition	Subsidence	Decomposition /Subsidence	Ash (0-50cm)		Peat carbon (0-50cm)		Bulk density (0-50cm)		Carbon in top 0-1cm peat layer	Decomposability *
	$tC\ ha^{-1}\ y^{-1}$	$cm\ y^{-1}$	$tC\ ha^{-1}\ cm^{-1}$	$g\ kg^{-1}$	SD	$g\ kg^{-1}$	SD	$g\ cm^{-3}$		$tC\ ha^{-1}\ cm^{-1}$	%
Forest(FT)	9.76	2.22	4.40	4.85	0.15	622	29	0.151	1)	9.37	47
								0.118	2)	7.36	60
								0.183	3)	11.38	39
Re-growing forest (RF)	6.07	0.90	6.77	5.15	1.57	611	20	0.180	1)	10.98	62
								0.140	2)	8.54	79
								0.220	3)	13.41	50
Crop land (KV)	25.49	4.07	6.27	5.91	1.50	604	3	0.170	1)	10.28	61
								0.125	2)	7.57	83
								0.215	3)	12.98	48

1)Average bulk density, 2)Bulk density -SD, 3) Bulk density +SD;
* (Decomposition /Subsidence) / (Carbon in top 0-1cm peat layer)

CO_2 emission increased with fall of ground water table. Regression analysis for the relationship between peat decomposition and ground water table depth was conducted using the data compiled by Hooijer *et al.* (2006) together with the data obtained in this study. The results revealed that peat decomposition was significantly correlated to ground water table depth for the data set dived into two groups, land with woody vegetation (P<0.01) and without woody vegetation (P<0.01). The slope of the regression equation for the land without woody vegetation was higher than that for the land with woody vegetation. This indicates that peat soils in cropland are more decomposable than in forests (Figure 2). Murayama and Bakar (1996) showed that for peat decomposition increased with

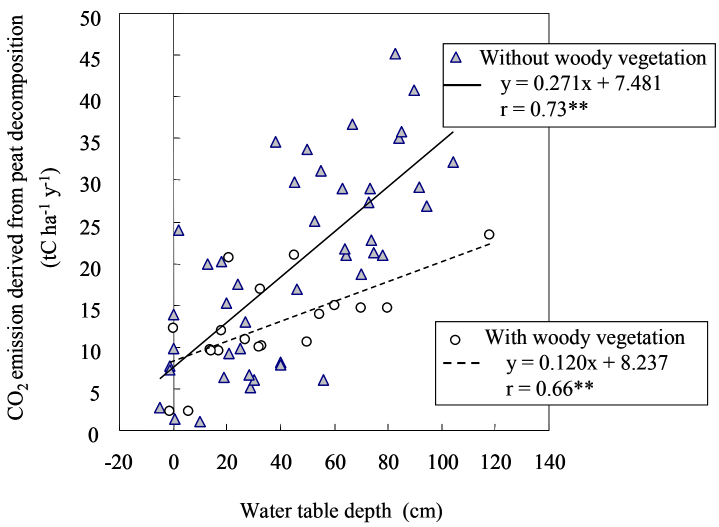

Figure 2. Relationship between annual mean water table depth and CO_2 emission derived from peat decomposition.

increase of soil pH and ash content of peat. Ash contents were highest KV and smallest in FT (Table 1). Farmers ordinarily apply ash to their fields in order to increase base cation content and soil pH of ombrotrophic acidic tropical peat soils, and peat fire supplies ash. These activities may stimulate peat decomposition.

N_2O emission

There was a significant exponential relationship between N_2O emission and peat decomposition ($P<0.01$) (Figure 3). The data includes previously reported values measured in a forest, oil palm plantation, sago palm plantation in Sarawaku, Malaysia (Melling *et al.* 2005; Melling *et al.* 2007). There was a large scatter in the relationship and there was a very wide range of N_2O emissions (from 0.04 to 4.39 kg N/ha/y in forests, re-growing forest and plantation and from 5.78 to 679 kgN/ha/y in croplands). Cropland enhanced N_2O emission significantly. Takakai *et al.* (2006) showed that annual N_2O emission increased with an increase of annual precipitation and N_2O was emitted proportionally to NO_3-N content in peat soil when peat soil became wet during the period of rainy season. This suggests that the N_2O was produced through denitrification in rainy the season using NO_3-N accumulated in peat soil through nitrification after mineralization of N released with peat decomposition in the dry season. Peat decomposition can involve heterotrophic respiration using both O_2 molecule and oxygen contained in NO_3^-(Hashidoko *et al.* 2007). Therefore, the large scatter in the relationship between N_2O and CO_2 emission is ascribed mainly to variation of nitrification and denitrification activities in soil which are influenced by the soil moisture condition and nitrate content

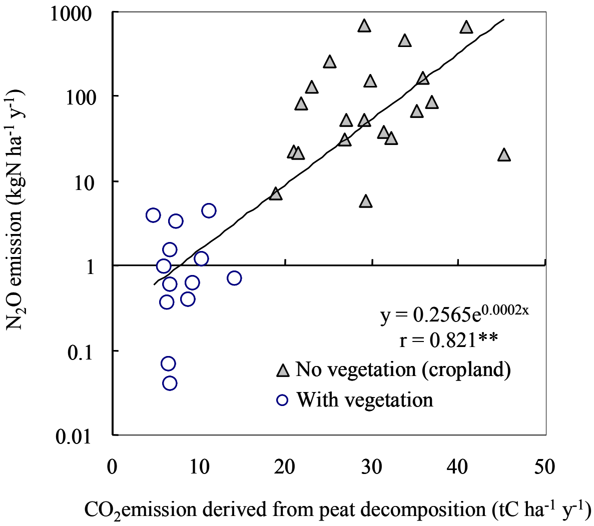

Figure 3. Relationship between CO_2 emission and N_2O emission

Conclusions

The fall of ground water table depth associated with drainage and agricultural land use in peatlands triggers the increase of peat decomposition. The higher decomposability of cropland peat might be caused by an increase in ash content of the peat soil. N_2O emission was also higher in cropland during the rainy season due to the high activity of denitrification.

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Carbon fractions and enzymatic activities in two cultivated dryland soils under conservation tillage

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Abstract
Long-term soil management experiments are expected to provide important information regarding sustainable crop production systems. In this study we evaluated the long-term effect of conservation tillage (CT) on biological properties in two different textured soils [Entisol (soil A) and Vertisol (soil B)].The results were compared with those obtained under traditional tillage (TT). Soil labile carbon fraction was measured by the determination of active carbon (AC) and water soluble carbon (WSC). Biological status was evaluated by the measurement of soil microbial biomass carbon (MBC) and enzymatic activities [dehydrogenase activity (DHA) and β-glucosidase activity (β-glu)].Contents of AC, MBC, and β-glu in soil A and contents of DHA in soil B were higher in CT than in TT, at 0-5 cm depth. Discriminant analysis (DA) showed that the discriminant function was strongly correlated with MBC and AC in soil A and with TOC, AC and WSC in soil B. According to this study, AC content was the most sensible and reliable indicator for assessing the impact of different soil management strategies on soil quality in the two soils. Long-term dryland soil conservation management improved the quality of both soil types (Entisol and Vertisol), especially near the surface, by enhancing its biological status.

Key Words
Permanganate oxidizable carbon; microbial biomass carbon (MBC) to TOC ratio, dehydrogenase activity, β-glucosidase activity.

Introduction
Conservation agriculture attaches great importance to maintain soil structure, productivity and biodiversity. Conservation tillage reduces the negative impacts of tillage, preserves soil resources and could lead to accrual of the soil carbon lost during tillage (Conant *et al.* 2007). Organic matter is involved in the enhancement of soil quality since it acts on soil structure, nutrient storage and biological activity. Small changes in TOC resulting from changes in soil management are often difficult to measure and several years are required to detect changes resulting from management practices. However, changes in small but relatively labile fractions of soil organic carbon may provide an early indication of improvement in response to management practices. The labile fractions of soil organic carbon are important to study as these fractions of soil C are more easily available carbon sources for soil microorganisms, and therefore influence nutrient cycles and many biologically-related soil properties. The labile fractions of soil C are often called the active C pool to distinguish from the passive C pool that is only very slowly altered by microbial activities. Active C (AC) is more related to microbial activity and more sensitive to soil management than TOC (Weil *et al.* 2003). Fractions of soil organic carbon that represent active C include microbial biomass C (MBC), particulate organic matter and sugars. Soil microbial properties such as MBC and activity of microbial populations and soil enzymes have been used to predict soil biological status and the effect of farm management as it relates to soil quality (Melero *et al.* 2007). The main aim of this work was to asses the long-term effect of conservation tillage on soil organic C fractions and biological properties in two different textured soils (sandy clay loam and clay) under a Mediterranean dryland cropping system. We hypothesised that long-term conservation tillage may have a positive effect on soil quality by improving soil organic carbon sequestration and biochemical properties. Moreover, we considered these parameters as reliable indicators of changes in soils with a long history of conservation tillage.

Methods
Localization and management systems
A long-term soil management experiment was established in 1991 at the experimental farm of the Instituto

de Recursos Naturales y Agrobiología de Sevilla (IRNAS-CSIC) on a sandy clay loam soil (Soil A) classified as a Xerofluvent (Entisol), with a $CaCO_3$ content in the 25-28% range and a clay content of about 25%. The second experiment (Soil B) was carried out in a long-term soil management experiment established in 1982 at the “Tomejil” dryland farm in Carmona (Seville) on a clay soil classified as a Chromic Haploxeret (Vertisol), containing 60% clay.

Both experiments were carried out using completely randomised designs (three replicates per treatment) and were conducted in 6 subplots of 300 m^2. Two treatments were tested: traditional tillage (TT) and conservation tillage (CT). In both trials, TT consisted of mouldboard ploughing (aprox 30 cm depth). The CT in soil A was characterized by leaving the crop residues on the soil surface, chiselling 25-30 cm depth every two years and a yearly disk harrowing to 5-7 cm depth. In soil B, CT was only characterized by leaving the crop residues on the soil surface. A wheat–sunflower-legume crop rotation was established for both treatments. The soil was fertilized in the same way in both treatments, following the regular practices of the local farmers. A basal dressing of 400 kg of a compound fertilizer (15N–15P_2O_5–15K_2O) (Soil A) and of 220 kg /ha (18N-46 P_2O_5-0 K_2O) more a top dressing of 200 kg /ha of urea (Soil B) were applied during the cereal season. Weeds were controlled in TT by tillage and in CT by the application of glyphosate.

Sampling and soil chemical and biochemical analysis

In both experiments, soil samples were collected on the same day at three depths: 0-5, 5-10 and 10-20 cm. Soil sampling was done four months after sowing a pea crop in soil A and a wheat crop in soil B in March 2008. Three soil cores of each individual subplot were taken. The moist field soil was sieved (2 mm) and was homogeneized and subdivided in two subsamples. One of them was immediately stored at 4ºC for microbiological and enzymatic activities and the other one were air-dried for chemical analysis. In air-dried subsamples, TOC was analysed using the Walkley and Black (1934) method. WSC was determined in a (1/10) aqueous extract using a TOC V-CSH Shimadzu analyser, AC according to Weil *et al.* (2003) and MBC content by the method modified by Gregorich *et al.* (1990). DHA and β-glu activities were determined according to Alef and Nannipieri (1995).

Statistical analysis

Statistical analyses were carried out using SPSS 11.0 for Windows and the results were expressed as mean values. Significant differences between management systems (TT, CT) were shown by a Student’s t-test at $p<0.05$. The data set of soil chemical and biochemical variables was also analysed using discriminant analysis (DA).

Results and Discussion

In soil A, CT presented higher AC, WSC and MBC contents than TT at 0-5 and 5-10 cm depth but only AC and MBC mean values were statistically different between both treatments (Table 1). In general β-glucosidase activity values were higher under CT than under TT at all studied depths, although only significant differences were observed in β-glucosidase activity at the 10-20 cm depth (Table 1). In soil B, TOC, AC and WSC contents were higher in soils under CT than soil under TT at 0-5 and 5-10 cm depths (Table 1). However, statistical differences between treatments (TT and CT) were only found in TOC and AC values at the superficial layers (Table 1). Values for DHA, in soil B, were higher under CT at 0-5 and 5-10 cm depths, although significant differences were only observed between treatments at 0-5 cm depth (Table 1).

Graphical representation in one dimension of the discriminant analysis of soil chemical and biochemical properties under TT and CT is show in Figure 1. The discriminant analysis showed that MBC and AC in soil A and TOC, AC and WSC in soil B were significantly different between TT and CT (Table 2). Conservation tillage systems have been shown to maintain and/or increase soil organic matter at higher levels than traditional tillage systems (Chivenge *et al.* 2007; Madejón *et al.* 2007; Melero *et al.* 2008). Under conservation tillage systems, the absence/reduction of soil disturbance produces a modification of surface soil conditions, which improves soil physical properties and reduces soil organic matter decomposition. Franzluebbers *et al.* (1995) reported that active fractions of soil organic matter increased in superficial soil layers in no tillage compared to traditional tillage

The increase in TOC, AC, WSH and MBC at the surface in CT in both experiments may also be associated with the high input of crop residues left on the soil surface and with their slower decomposition processes (Salinas-Garcia *et al.* 2002). Differences between the two soils were also observed. The amount of TOC loss

Table 1. Mean values ± standard errors of soil chemical and biological properties in soil A and B under traditional tillage (TT) and conservation tillage (CT). Differences between treatments for each depth from a t-student are indicated by (*) (p< 0.05).

	Treatment	SOIL A Depth (cm)			SOIL B Depth (cm)		
		0-5	5-10	10-20	0-5	5-10	10-20
TOC	TT	9.84± 1	9.03± 1	8.23± 1.6	8.37± 0.2	7.60± 0.2	6.57± 0.8
(g /kg)	CT	10.85± 0.7	8.87± 0.3	8.65± 0.7	11.53*± 0.2	9.70*± 0.1	8.23± 0.9
AC	TT	780± 49	694± 0.9	695± 1.5	697± 1.2	695± 0.0	695± 2
(mg/kg)	CT	1680*± 205	1039*± 19	693± 3.1	708 *± 3.2	702± 2.0	694± 1.2
WSC	TT	60.4± 5.2	50.7± 1.6	42.4± 0.01	46.0± 3.6	38.8± 2.4	35.4± 2.3
(mg/kg)	CT	82.9± 24	65.6± 8.1	46.6± 3.2	112 ± 40	87.6± 30	71± 36
MBC	TT	814± 42	806± 40	780± 53	378 ± 26	357± 13	314± 33
(mg/kg)	CT	1058*± 32	978*± 52	879± 25	766 ± 274	304± 0.2	272± 67
DHA	TT	1.16± 0.3	0.66± 0.2	0.49± 0.2	1.06 ± 0.1	0.64± 0.1	0.56± 0.2
(mg TPF /kg)	CT	1.15± 0.4	0.72± 0.2	0.26± 0.2	3.04*± 0.2	0.94± 0.1	0.29± 0.1
β-Glu	TT	140± 19	84.2± 11	55.6± 11	122 ± 24	63.8± 21	43.1± 11
(mgPNP /kg/h)	CT	169± 19	108± 13	98.8*± 5	128 ± 36	45.4± 14	32.2± 9.2

Table 2. Correlation (in order of importance) of each variable to the discriminant function in both soils.

SOIL A		SOIL B	
	Structure matrix Function 1		Structure matrix Function 1
MBC	-0.607	TOC	0.537
AC	-0.373	WSC	-0.406
WSC	-0.202	AC	0.340
TOC	-0.072	DHA	0.236
DHA	0.032	MBC	0.129
β-*Glu*	-0.024	β-Glu	-0.048

TOC: Total organic carbon; AC: active carbon;
WSC: water soluble carbon; MBC: microbial biomass carbon;
DHA: dehydrogenase activity; β-Glu: glucosidase activity.

due to tillage is dependent on the clay content of the soil. In general, higher TOC losses are observed in coarse textured than in fine textured soils (Chivenge *et al.* 2007). Thus, we would have expected to obtain greater values for soil chemical and microbiological properties in the clay soil (soil B). However, in general the values were slightly higher in soil A. These results can be explained by the crop type growing at the sampling time [a pea crop (leguminous) in soil A and a wheat crop (gramineous) in soil B], rather than by soil characteristics. The rhizosphere of leguminous crops may more actively secrete higher amounts of exudates than wheat crops, and may supply therefore more carbon and nitrogen to the soil for succeeding crops than non-legume crops (Nuruzzaman *et al.* 2005).

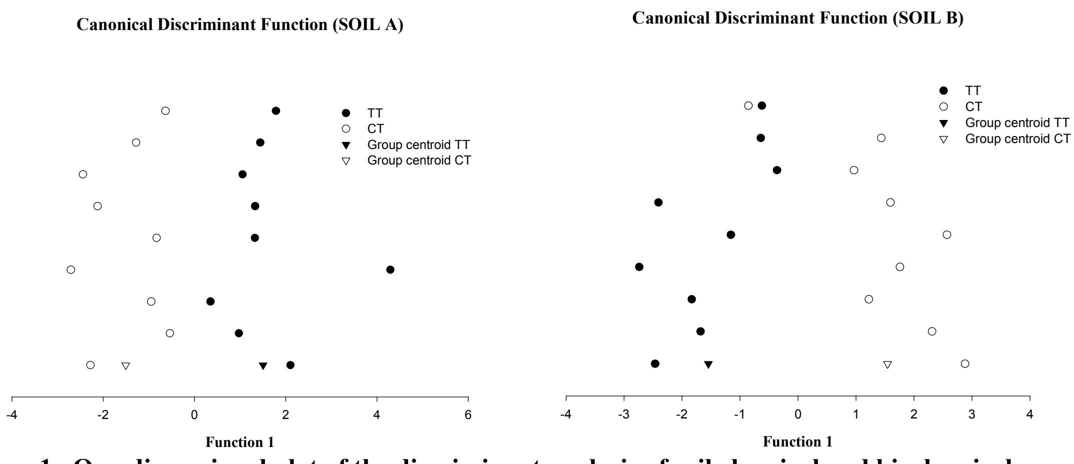

Figure 1. One-dimensional plot of the discriminant analysis of soil chemical and biochemical properties from soil under traditional tillage (TT) and conservation tillage (CT).

In both soils, CT had a positive effect on microbial biomass and enzymatic activities. Soils under CT have been shown to have higher soil microbial biomass and enzymatic activity values than those under conventional tillage systems (Madejón *et al.* 2007; Melero *et al.* 2007), indicating an activation of microorganisms through carbon source inputs of organic residues. The results of the discriminant analysis showed that AC content was the parameter that best explained the total variance of the data set in both soil types, revealing the advantages of conservation tillage. Although WSC content is more commonly used as the soil quality indicator in agroecosystems (Madejón *et al.* 2007), this study showed that AC content is the most sensitive and reliable indicator for assessing the impact of different soil management systems on soil quality.

Conclusions

Long-term soil conservation management improved the quality of both soils (Entisol and Vertisol) through enhancing the organic carbon fraction and biological status, especially near the surface. Enhancement of soil quality may be more related to the crop type existing at the sampling time than to differences in soil texture. In both soils, AC content was the best indicator of the total variance of the data set for assessing the differences between tillage on soil quality.

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Changes in carbon and nitrogen stocks following conversion of plantation forest to dairy pasture on Vitrands (Pumice Soils), New Zealand

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Introduction

Between 1990 and 2010 some New Zealand plantation forests underwent deforestation to establish dairy farms. The main area of land-use conversion to pasture is to the north of Lake Taupo in the Central North Island (Figures 1 and 2). *Pinus radiata* (radiata pine) plantations were established in the late 1920s-early 1930s because the Vitrands (Pumice Soils) predominant in the Central North Island were deficient in Co and other trace elements, causing a fatal stock disease in sheep and cattle known as 'bush sickness'. Bush sickness was subsequently rectified in the mid-1930s with the regular addition of Co, so pastoral farming became viable. The high price of milk solids has recently led to renewed interest in dairying. Recent studies have shown carbon can accumulate following deforestation and establishment of pasture (Fearnside and Barbosa, 1998; Murty *et al.* 2002; Hedley *et al.* 2009). However, more information on the rate of accumulation of carbon after deforestation is needed. Increases in soil carbon can improve physical and chemical soil properties, and is an important store of global carbon.

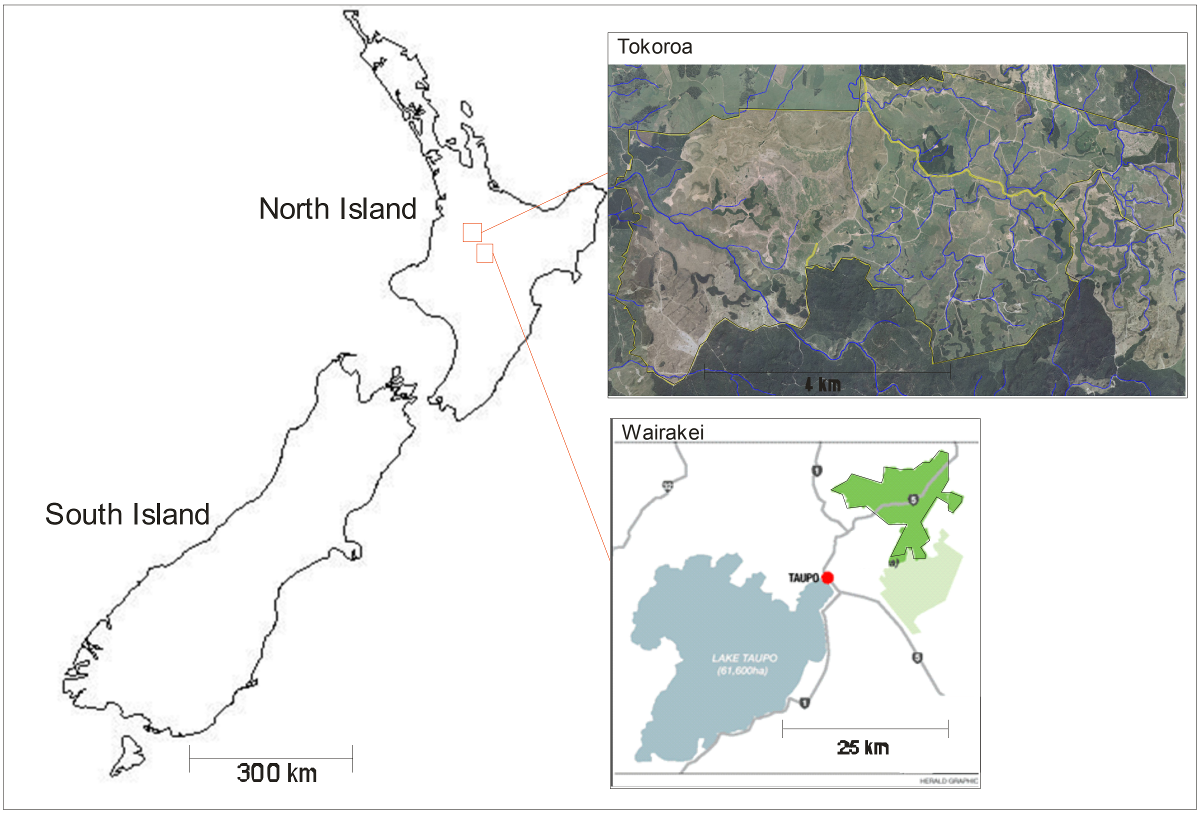

Figure 1. Location of study area, central North Island, New Zealand.

Hedley *et al.* (2009) found a mean rate of 4.07 mg cm^{-3} per year carbon accumulation in the central North Island at sites that had undergone deforestation and conversion to pasture. They sampled the top 150 mm of soils, at sites 1 and 5 years after conversion, as well as sites in pasture for over 20 years. The highest concentration of carbons was reported in long-term pastures (Hedley *et al.* 2009).

The objective of our study is to determine the rate and magnitude of change in soil carbon and nitrogen following conversion to pasture from plantation forest. Specific objectives include determining the carbon and nitrogen concentration for the soil profile down to 60 cm along with the change in carbon to nitrogen ratio with a change in land use.

Figure 2. Pasture establishment following deforestation, approximately 10 km north of Taupo, central North Island, New Zealand (a. land being prepared for pasture, b. land 3 years after pasture establishment).

Study sites and methods

My study examines two study areas in the Central North Island, one near Taupo and the other about 50 km to the north, near Tokoroa (Figure 1). Both areas have undergone conversions from second or third rotation *Pinus radiata* forest to dairy pasture. Soils are Udivitrands (Soil Taxonomy) or Pumice Soils (NZ Soil Classification) formed on the Taupo tephra deposited in 232 ± 4 AD, i.e., about 1780 years ago (Hogg *et al.* 2009), and are on either flat or rolling land under similar humid, temperate climates (udic moisture and mesic temperature regimes). Sites ranging from current *Pinus radiata* forest through sites that have been under dairy pasture for 2, 3, 4 and 5 years to over 50 years under dairy pasture have been identified at each study area, giving 14 sampling sites. All sites are on the same landscape unit, a terrace with an elevation of 300 – 400 m asl. Three soil pits were excavated and sampled at each site. Soil bulk densities were measured using 6 cm diameter and 5 cm deep rings. Samples were oven dried and weighed to determine the dry bulk density of soil. Soil carbon and nitrogen content is to be determined on bulk samples of soil taken from individual horizons from each soil pit, using an emission on combustion method. To capture some of the variability a 60 cm corer was used to take 18 cores from around each paddock in which the pits have been dug. Each core was split into horizons and the horizons bulked for a carbon and nitrogen sample. Samples will be taken down to 60cm where possible.

Preliminary results and discussion

There are distinct pedological differences between the soils under forest, recently converted sites, and longer term pasture. Soil bulk densities in the A horizon are lowest under forest. Soil A horizons are more strongly developed with stronger aggregate development under long-term pasture. Soil dry bulk densities for a site 2 years since pasture establishment had an Ap horizon ranging from 0.62 to 0.72 g cm^{-3}, Bw were 0.70 to 0.82 g cm^{-3}, and Cu were between 0.74 to 0.77 g cm^{-3}. Below 50 cm a paleosol was found. The paleosol had the most consistent results for bulk density with between 0.63 and 0.64 g cm^{-3}. Further analytical work is being undertaken at multiple sites to obtain bulk densities along with carbon and nitrogen concentrations.

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Development of a soil carbon benchmark matrix for central west NSW

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Abstract
The development of a spatial framework for soil carbon data is described. The theoretical basis is a general conceptual model for soil carbon in combination with the general soil carbon equation used in most soil carbon models. A general set of soil, climate, land management and landform factors that influence soil carbon levels are described. How existing sources of information including soil maps, climate data and land management data can be used to develop a soil carbon matrix is briefly described. Some of the advantages of developing this soil carbon matrix are explained.

Key Words
Spatial framework, soil carbon potential, land management.

Introduction
Soil carbon is the prime determinant of biological soil condition and globally is the largest component of terrestrial carbon. Soil carbon is impacted by climate, soil type, land use and land management practices. Practices that increase soil carbon improve soil condition and remove atmospheric carbon and soil carbon storage across NSW landscapes have the potential to contribute significantly to climate change mitigation and climate change adaptation. However, there is an urgent need to interpret and combine existing data into a comprehensive spatial framework to track and predict soil carbon levels across the State. This spatial framework can be based on a matrix of soil x climate x land use x land management combinations that are representative of each region. This soil carbon matrix will provide a mechanism for assessing the impacts of land use and land management change across regions. It will provide capacity to rapidly estimate expected critical or benchmark soil carbon levels for different locations throughout NSW. The matrix will potentially provide a capacity to apply Market Base Instruments as a natural resource management tool and even possibly a tool for soil carbon trading. The difficulty is in populating the soil carbon matrix. This paper outlines some of the problems and possible methodologies that can be applied to develop such a soil carbon matrix for two catchments in central west NSW.

Developing the spatial matrix for soil carbon - theoretical basis
Soil organic carbon (SOC) levels in soils can be described broadly by the following set of equations.

$$\text{SOC} = f\,(\text{soil, climate, land use/land management practice, landform, time}) \qquad 1$$

which is a conceptual model.
In addition use can be made of the general soil carbon equation (Dalal and Chan 2001).

$$\text{SOM}t = \text{SOM}_0 \exp(-kt) + \text{A}/k\,[1 - \exp(-kt)] \qquad 2.1$$

Where SOM_0 and SOM_t are the SOM contents initially ($t = 0$), and at a given time, t, A (mass of SOM per unit area) is the rate at which organic matter is returned to the soil; and k (reciprocal of time) is the rate of loss of SOM or the rate of decomposition. The value of k will vary with the nature of the organic matter and the amount of soil carbon in each of the carbon pools or fractions.
The values of A and k will vary with the range of factors as described in Table 1. The soil and climate factors will affect the values of A and k in these equations. Position in the landscape will also affect A and k by affecting the accumulation and redistribution of water, nutrients, sediments and organic matter. Soils on ridges and upper slopes will tend to lose soil and organic matter which will tend to accumulate on lower slopes and in depressions. Generally soils in lower slope positions will tend to have a wetter moisture regime for longer. In turn land management practices have a very large impact on A and k and can be used to manipulate A and k to bring about changes in soil organic matter and soil carbon.

Soil organic matter is heterogeneous and composed of different fractions or pools. A more accurate description of changes in soil carbon is then given by:

$$\mathrm{SOM}t = \mathrm{SOM}_1 \exp(-k_1 t) + A_1/k_1 [1 - \exp(-k_1 t)] + \ldots\ldots\ldots\ldots + \ldots\ldots\ldots + \mathrm{SOM}_n \exp(-k_n t) + An/k_n [1 - \exp(-k_n t)] \quad \ldots\ldots\ldots 2.2$$

Where the terms represent the amount of soil organic matter in a given fraction in the soil (SOM_n), the amount of organic matter in a given fraction added to the soil (A_n) and the decomposition constants for the fractions k_n.

Land management practices can change the amount of soil carbon in each of the pools or fractions. For example it is likely that organic matter derived from trees and woody native vegetation is likely to have more soil carbon in the more resistant soil carbon fractions. There are some suggestions that this also occurs with organic matter derived from perennial grasses but this is currently under investigation.

Equation 1 can be used to estimate how the values of SOM_0, A and k will vary across the landscape with soil type, climate and changes in land management practice and land management activities. Table 1 provides a list of the factors that can influence the values of A and k in equations 2.1 and 2.2.

Developing the spatial matrix for soil carbon - steps

The steps to developing a spatial matrix for soil carbon will include the following.

A. Determining combinations of soils and climate that have an equivalent soil carbon potential. These combinations can be termed soil carbon zones for convenience and within these specified land management practices will have a defined soil carbon potential. For practical purposes, the soil carbon potential is defined as the amount of soil carbon that can be accumulated to 30 cm at long term equilibrium under a specific land management practice. In defining these combinations some initial assumptions will be made based on the factors in Table 1.

B. Using existing soil landscape maps and climate information (SILO Database and FullCAM climate data) the soil carbon zones will be spatially delineated using GIS layers. The existing soil landscape maps can potentially provide the soil information identified in Table 1, while the climate data sources can easily provide the climatic data. The land management data required for each region is more difficult to obtain. Use can be made of the review by Valzano *et al.* (2005). Several monitoring programs and good local knowledge from local advisory and land resource officers will be of great value in obtaining useable land management data.

C. The soil carbon potentials for different land management practices within the soil carbon zones will be estimated using data from a range of sources including:

a. Experimental and high quality monitoring soil carbon data (DECCW MER Program), including reviews (Valzano *et al.* 2005; Murphy *et al.* 2003).

b. Soil carbon data from routine soil testing and general monitoring soil carbon data (CMA and landholder data)

c. Modelling of soil expected carbon levels (FullCAM).

d. General expert knowledge estimates.

The outcome will be to populate a spatial matrix of benchmark soil carbon values that can be used as a basis for evaluating the soil carbon status of soils across the region. An example of how the results can be presented is shown in Figure 1 for one soil carbon zone.

Table 1. List of factors that influence the values of A and k in Equations 2 and 3. These factors will affect the rates of addition of soil carbon to the soil and the rates of decomposition of soil carbon. These factors will also influence how soil carbon is distributed between the different soil carbon fractions in the soil.

Soil factors /landform	Climatic factors	Land use and land management practices	
Landform Accumulation of soil carbon in lower slopes; More moisture in lower slopes; Deeper soils on lower slopes; Clay content / texture Soil nutrient levels Nitrogen; pH – soil acidity; Alkalinity; Phosphorus; Sulfur; Soil structural condition Water holding capacity; Surface soil sodicity; Subsoil sodicity; Compaction; Erosion; Soil depth Profile drainage Periods of free water; Periods of anaerobic conditions; Salinity Subsoil salinity; Surface salinity; Scalding.	Moisture regime Rainfall; Evaporation; rainfall /evaporation ratio; Temperature regime Annual average minimum temperatures; Annual average maximum temperatures; Temperature distribution through the year; Wind Wind erosivity;	Biomass production Fertilisation and agronomic management which can increase biomass production; more plant residue; Nutrient management critical, also acidification;, grazing management critical; Soil cultivation and tillage Results in greater soil aeration, loss of moisture; More rapid decomposition; decreased organic matter levels; Aggressive tillage exposes "protected" soil carbon; Lengthy fallow periods reduce plant biomass input to soils and therefore reduce overall organic matter levels; Burning – stubble or residues Releases the carbon stored in these residues; Oxidises carbon stored in the soil fire though can also add resistant char to the soil; Erosion from bare soil surfaces Can remove organic matter in water- or wind-borne soil particles; No-till, direct drill cropping practices (conservation type) – Has been shown to increase soil carbon by modest amounts; Needs to be part of a complete crop management package with fertiliser use, controlled traffic etc; Many other agronomic, economic and soil condition benefits.	Perennials pastures – Re-establishment and appropriate management of perennial components of farming systems can also enhance soil organic matter content; Re-establishment of perennial pastures (even as a component of a farming rotation) has been shown to increase soil organic matter in this environment more than any other farming technique; Careful management of these pastures can greatly enhance soil organic matter contents; Trees, native vegetation Soil organic matter is invariably higher under trees in this environment compared with other land-uses; Retaining or re-establishing trees in the region should be encouraged although trade-offs with agricultural production must be considered; Applications of manures, organic materials, composts, biochar These have been shown to significantly increase soil carbon. However, although on small horticultural plots, these applications might be an economic alternative, they usually require large quantities of application and this might be of limited utility in broad-acre farming.

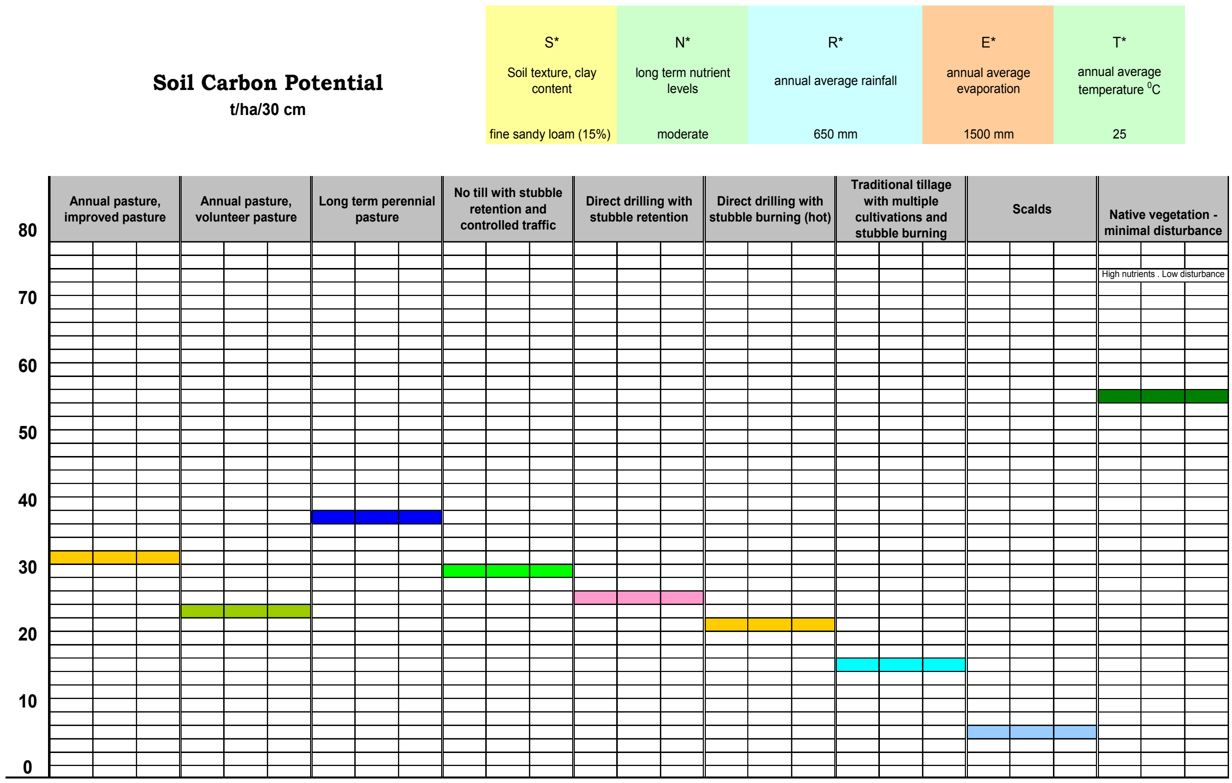

Figure 1. Possible representation of the soil carbon potentials for a range of land management options for a single soil carbon zone in Central West NSW. This is based on some soil carbon data for the Red Chromosol cropping belt within the Central West.

Conclusion

The concept of the soil carbon benchmark matrix is a useful tool to provide a spatial framework to organise existing and future soil carbon data. It has a sound theoretical basis and a range of practical applications for understanding the science and management of soil carbon.

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Development of a soil quality decision support tool to identify and support best management practices

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Abstract

Understanding and managing the effects of soil and crop management practices on soil quality and crop production is essential to maintaining profitable and environmentally sustainable farming enterprises. The Land Management Index (LMI) decision support tool was developed to assist land managers predict the effects of current management practices on soil quality and allows them to run "what if?" scenarios to support future paddock-scale management decisions. To achieve this, a total 14 different indicators of soil quality were measured on >700 paddocks representing the major soils orders and agricultural land uses across seven New Zealand regions. Detailed soil and crop management information (e.g. crop type, sowing/harvest dates, tillage type, irrigation) was also collected for the 10 years proceeding sampling for each paddock. The model was developed by first defining a minimum dataset of critical soil quality indicators and then establishing empirical relationships between these indicators and quantitative measures of the soil and crop management information. These empirical relationships formed the basis of the first generation LMI (ver 1.1) decision support tool that was released for use by farmers and resource managers in June 2009.

Key Words

Crop rotation, tillage, land use, aggregate stability.

Introduction

The soil and crop management practices applied to paddocks can be important determinants of changes in soil quality and the future productivity of agricultural production systems. Knowledge of these effects is important to establishing recommended best management practices and the development of tools that assist farmers and resource managers with on-farm management decisions. The ability to predict changes in soil quality and, by association, future productivity based on historic management information would be useful in undertaking "virtual" monitoring of soil quality changes and identifying real time management solutions to existing soil quality conditions based on modeled future "what-if" scenarios. The aim of this research was to develop and test methods for quantifying the effects of management history on soil quality properties across a wide range of soils and land uses in New Zealand and apply this knowledge to the development of a first generation soil management decision support tool.

Materials and methods

Soil quality and management history data were collected from >700 paddocks between June 2002 and July 2007. Data from a further 37 paddocks was also collected as an independent validation dataset. The paddocks sampled include a range of land uses and intensities (mixed and intensive arable and vegetable cropping, dairy pasture, intensive beef pasture and sheep pasture) and were located across seven New Zealand regions (Auckland, Waikato, Gisborne, Hawke's Bay, Manawatu, Canterbury and Southland). For each paddock, detailed soil and crop management information was collected, including crop types and rotation, sowing and harvest dates, residue management practices, individual tillage passes (e.g. mouldboard plough, harrow, roll), irrigation and fertiliser inputs, for the 10 years prior to soil quality assessment. No two paddocks monitored had exactly the same management over the 10 year history period. Much of the management history information was descriptive by nature and hence needed to be converted to a quantitative form before it could be used to derive coefficients for the LMI model. This was achieved by applying quantitative weightings to the individual tillage implements used to prepare seed beds and the crop types grown. A time weighting was also applied to account for the effects of time over the 10 years preceding each soil quality assessment. The details of these analyses and outcomes are discussed below.

Fourteen different indicators of soil quality were measured on each paddock, covering physical, chemical and biological aspects of soil quality. These included: total C and N (%, t/ha, 0-15 and 15-30 cm), hot-water extractable carbon (HWC)(µg C/g soil, 0-15 cm), C:N ratio (0-15 and 15-30 cm), bulk density (BD)(g/cm^3, 0-15 and 15-30 cm), penetration resistance (PR)(MPa, 0-15 and 15-25 cm), aggregate stability (MWD; mm,

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%<1 mm, 0-15cm), aggregate size distribution (MWD mm, 0-7.5 cm), erodible aggregates (%<0.85 mm diameter, 0-7.5cm), large-dense aggregates (%>9.5 mm diameter, 0-7.5 cm), ideal aggregates (%0.85-9.5 mm diameter, 0-7.5 cm), Olsen P (µg /g, 0-15 cm), pH (0-15 cm) and soil texture (0-15 cm). Further supporting information was also collected for each paddock including soil series, soil order, soil texture, and the Land Environments New Zealand (LENZ) Level 1 climate layer descriptors (Leathwick et al, 2003). LENZ Level 1 descriptor is an environmental classification, which groups together those sites with similar environmental conditions). These factors (soil order, texture, LENZ level 1) are referred to here as the site/location factors.

Principal components analysis (PCA) and correlation analyses were applied to the indicator data to define a minimum dataset of critical indicators for use in development of the LMI decision support tool. Our aim was to 1) quantify the contribution of different indicators to explaining variability in the indicator dataset, 2) determine the inter-relatedness of different indicators, and 3) ensure that the indicators selected address a wide range of key soil management issues (e.g. soil organic matter storage, nutrient content, soil structure, soil compaction).

Results and discussion

Selecting a minimum dataset of critical indicators

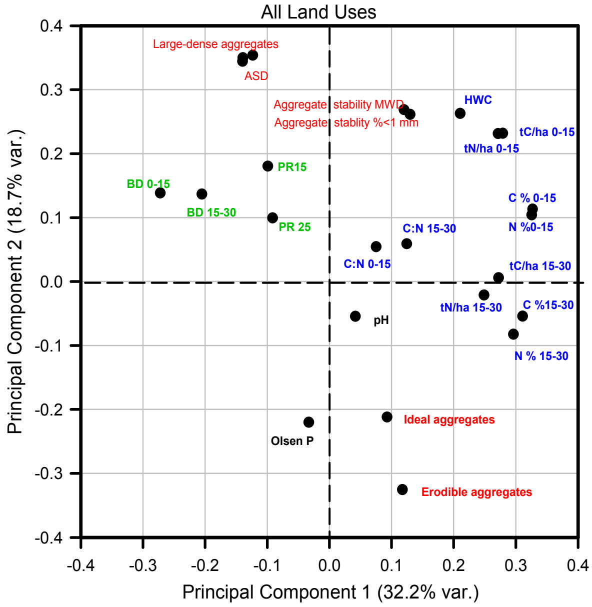
The latent vector loadings produced by the PCA analysis provided a measure of the contribution of each indicator to each PC axis. The aim was to identify indicators that make a relatively large contribution (i.e. have a high [+ or –] latent vector loading) to those principal components that explain a relatively high percentage (e.g. 70% or more) of the variation in the indicator data. Correlation matrices were also used to identify those indicators that are quantitatively related and therefore are likely to provide similar information about the state of the soil. The vector loadings for the first two principal components of the data set are plotted in Figure 1. The indicators with the highest loadings are shown as those points plotted the furthest from the centre point along one or both the PC axes. Just over 50% of the variation in the entire data set was explained by the first two PCs with a further 20% explained by PCs 3 and 4, and a total of 95% explained by the first 10 PCs. The indicators with the highest loadings to PC 1 (32.2% of the variation) were dominated by those that describe soil organic matter (e.g. total C and N % and t/ha, 0-15 and 15-30 cm) but also included biologically active carbon (HWC) and compaction (BD 0-15 and 15-30cm). The positive loadings for soil organic matter indicators and the negative loading for bulk density indicators are consistent with their established inverse relationship, i.e. bulk density tends to decrease with increases in soil organic matter content. The indicators with the highest loadings to PC 2 (18.7% of variation) were dominated by those that describe soil structure (e.g. aggregate size distribution [%] and aggregate stability [MWD, mm]) but also included some indicators of organic matter in the surface soil (e.g. tC/ha 0-15cm), biological activity (e.g. HWC) and chemical fertility (e.g. Olsen P). Most of the soil structure indicators also contributed high loadings to PC 3 (12.1% of variation), as did indicators of compaction (i.e. BD and PR) and chemical fertility (e.g. Olsen P). As a result of these analyses the minimum dataset of critical indicators was defined as; total carbon (t/ha, 0-15 cm), HWC, aggregate stability, erodible aggregates, large-dense aggregates, Olsen P, PR and C:N ratio.

Quantifying soil and crop management information

Tillage is generally accepted to have a neutral to negative effects on soil physical quality. The tillage practice used to establish a seed bed for a given crop is often based on a combination of implements. The LMI management dataset included 47 different tillage implements combined in different ways. We adapted the method used in the US 'Soil Conditioning Index' (National Agronomy Manual 509) developed by the United States Department of Agriculture Natural Resources Conservation Service to convert descriptive data into quantitative data. Each tillage implement was assigned a soil disturbance score based on the degree of disturbance (none, 0, to high, -5) associated with inversion, mixing, lifting, shattering, aeration and compaction (Table 1). The scores for different actions were then summed and multiplied by the proportion of each paddock affected by the implement to give the Soil Disturbance Rating (SDR). Where several tillage implements were used to prepare the soil for sowing a single crop, the SDRs for individual implements were summed to calculate the Total SDR for each crop sown (Table 2).

Crops are generally considered to have a neutral to positive effect on soil physical quality, as they provide the soil with protection against erosion by wind and rain, add organic matter to soil through root turnover, and release root exudates that stimulate biological activity and help to form and stabilize soil aggregates.

However the contribution of different crops to soil quality differs greatly. Hence different crop types were assigned positive weightings based on published literature and expert opinions on crop rooting characteristics, organic matter returns and nitrogen fixation potential (Table 3).



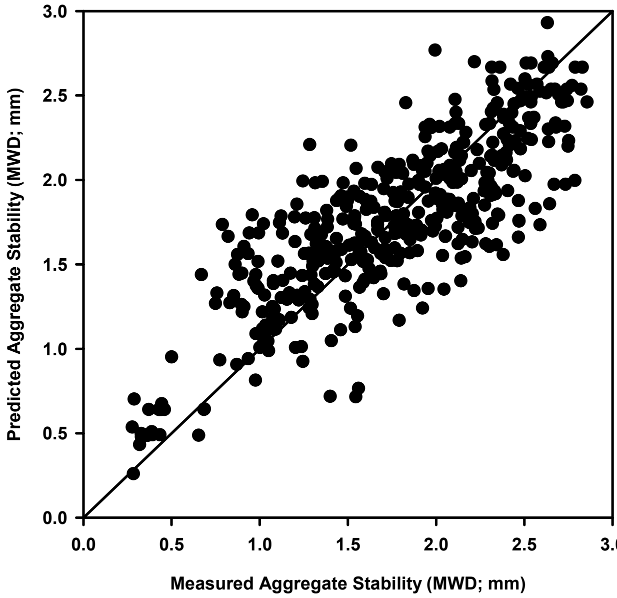


Figure 1. Vector loading for the first two principal components for the full LMI dataset.

Figure 2. LMI (version 1.1) predicted vs measured values of aggregate stability (MWD, mm) (454 paddocks).

Table 1. Examples of soil disturbance scores given to different tillage implements that were used to define their soil disturbance rating.

Tillage implement	Disturbance scores						Proportion of paddock affected	Soil Disturbance Rating
	Inversion	Mixing	Lifting	Shattering	Aeration	Compaction		
Direct drill	-1	-1	-2	-2	-1	-1	0.5	-4
Grubber	-4	-4	-3	-4	-4	-3	1	-22
Harrow	-2	-3	-1	-4	-3	-1	1	-14
Mouldboard plough	-5	-5	-5	-5	-5	-4	1	-29
Tyne	-1	-2	0	-2	-3	-3	1	-11

Table 2. Example calculation of the total soil disturbance rating (SDR) for a series of tillage passes used when establishing a barley crop.

	Pass 1	Pass 2	Pass 3	Pass 4	Pass 5	Total SDR
Implements used	Mouldboard plough	Maxi-till	Maxi-till	Harrow	Roll	
SDR of implement	-29	-18	-18	-14	-4	-83

Table 3. Examples of the crop type weightings applied.

Crop type group	Example	Weighting[1]
Fine root	Grass, triticale	4
Cereal	Barley, wheat	2.5
Brassica	Broccoli, pasja	2
Course root	Maize, sweetcorn	2
Tap root	Canola, mustard	2
Leafy vegetable	Lettuce, spinach	1
Root crop	Carrot, potato	1
Fallow/none	-	0

[1] values range from 0 to 4, with higher values having the greatest benefits on soil quality.

Recent management events have a stronger influence on soil quality than more distant events, hence a linear monthly time weighting was applied to all tillage SDR and crop weightings (based on 120 months). All tillage was assumed to occur in the month of crop sowing. All time-weighted tillage SDR values were summed to a single 'tillage score' and used in subsequent analysis, as were all time-weighted crop scores. Alternative time weighting relationships (exponential, logistic and Gompertz curves) were investigated, but the linear weighting was found to give the best fit to the data (results are not presented here).

Model development

The regression analysis procedures in GENSTAT were used to define the quantitative relationships between the site/location factors, the time-weighted tillage and crop scores and the soil quality indicators. Stepwise regression analysis was first used to identify which factors and variates contribute significantly and most importantly to describing variability in each indicator. Next multiple linear regression analysis was used to define the best case model and to establish how much of the variation was explained by the main effects and interactions (only first-order interactions were considered) that contributed significantly to the best fit model. Multiple linear regression was used to obtain the coefficients required to parameterise the LMI model.

Following this approach a first generation LMI (version 1.1) was developed based on data from arable and vegetable cropping and sheep pasture land uses (454 paddocks). The success of the method described for converting descriptive crop and tillage information to quantitative measures was determined by how much of the variability in certain soil quality indicators could be explained by the summarised management information. For most of the indicators, there was a relatively large and significant improvement in the amount of variability that could be explained by adding the time-weighted tillage and crop score information into the analysis. For example, in the case of aggregate stability, when location factors (texture, soil order, LENZ Level 1) alone were included in the regression analyses, only 20% of the variation in aggregate stability (MWD, mm) could be explained. When time-weighted crop and tillage scores were included in the regression analyses, along with the location factors, a further 44%, or a total of 64% of the variation was explained. The addition of the crop and tillage factors markedly improved the prediction of aggregate stability values for the cropping and pastoral land uses sampled. The relationship between measured aggregate stability and that predicted by the LMI model can be seen in Figure 2.

Conclusions

Our results indicate that the method used to quantify soil and crop management information can be useful in explaining variability in soil quality data and may be applied to the development of soil management decision support models operating at the paddock scale. Quantification of descriptive management information allows its incorporation in development of decision support tools. The LMI decision support tool allows land managers to predict soil quality following different management practices. This information will be useful in monitoring the effects of current land management practices on soil quality without the high cost of sampling and measuring changes in soil properties on paddocks over time. However, perhaps more importantly it will also help farmers and land managers to assess the likely effects of future management changes on soil quality helping support future on-farm management decisions. Further information on the LMI can be obtained from Plant and Food Research or the authors.

Acknowledgements

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Distribution of forms of soil potassium in the Central highlands of Papua New Guinea and its implications to subsistence sweet potato (*Ipomoea batatas*) production

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Abstract
Sweet potato being the important stable food crop of Central highlands of Papua New Guinea, it is essential to ascertain the soil potassium reserves in the gardens. Traditionally, sweet potato is cultivated without addition of any mineral K inputs. Soils from two depths (0-10cm) and (10-20 cm) in two types of gardens (old and new gardens) were assessed for different fractions of soil potassium in volcanic and non volcanic soil groups. Water soluble K content constituted less than 1% of the total K. Exchangeable K content contributed to about 2.0- 3.1 % of the total K, while, non exchangeable K contents was in the range of 2.5-4.2 %. Both, exchangeable K and non exchangeable K were significantly lower, in volcanic soils and old gardens, compared to non volcanic soils and new gardens. Non exchangeable K content was low in more than 95% of the gardens. Especially, older gardens which are in volcanic soil groupings are more susceptible to the K depletion problem due to continuous sweet potato cultivation, possibly owing to their lower mineral K status. Such gardens should be managed either with allowing sufficient fallow period for regeneration of soil fertility or with suitable dose of mineral K fertilizers.

Key Words
Forms of potassium, exchangeable K, non exchangeable K, sweet potato gardens, potassium reserves, garden types.

Introduction
In the highlands of Papua New Guinea, sweet potato (*Ipomoea batatas L.)* is the major stable food crop (Bourke 1985). The population of the highlands region, however, has been increasing by 3% each year and placing increasing pressure on the land to produce extra food for the growing rural populace. Simultaneously, crop productivity appears to be declining in gardens and this decline has been attributed to a reduction in soil fertility linked to the progressive shortening of fallow rejuvenation periods (Allen *et al*. 1995; Sem 1996; Bourke 2005). Results from previous work conducted across four of the highlands provinces on soil and crop variables for old gardens (cultivated over many seasons) and new gardens (newly brought into cultivation) on soils of volcanic and non-volcanic origin suggests that K deficiency was the primary cause of poor crop production (Ramakrishna *et al.* 2009) in almost a third of sweet potato gardens, but was more of a problem in old gardens than in new (Bailey *et al.* 2009). Although these studies clearly identified K deficiency as the major cause of poor sweet potato production, further information on distribution and soil factors affecting K availability in sweet potato farming systems are not available. In particular, from a sustainability perspective, it is essential to ascertain if soil reserves alone are sufficiently large and sufficiently accessible to sustain sweet potato production in the medium to long-term (decades to centuries), in the absence of external K inputs (fertilisers) by simply relying on various improved fallow management practices.

Methods
Study sites and soil sampling
Forty sweet potato gardens, 15 in Eastern Highlands province, 4 in Simbu, 9 in Western Highlands and 12 in Enga, were selected for the purpose of K characterization. About half of the sites were identified on soils derived from volcanic materials while rest of them were developed from non- volcanic parent material. In each of the soil group, about half of the gardens were old (continuously cultivated gardens about to go into fallow), and about half were new gardens (recently reclaimed from fallow). At every site soil samples were collected at two depths; from surface (0-10 cm) and sub surface (10-20 cm). Further details on soil sampling protocols, processing of the soil for physicochemical analysis, and methods followed are given elsewhere (Ramakrishna *et al.* 2009).

Potassium characterization
Soil samples passing through 2mm mesh sieve was extracted for various K fractions. Water soluble K was extracted with deionized water (1: 5 w/v) after shaking for 30 minutes on a mechanical shaker and later contents were centrifuged to separate clear extract (Jackson 1973). Exchangeable K was determined by extracting the soil with neutral normal ammonium acetate, Non-exchangeable K was estimated as the difference between boiling 1N HNO_3 –K and neutral normal ammonium acetate K (De Turk et al 1943). Total K was estimated by the digestion of soil samples (ground to pass through 0.17 mm sieve) in HF-$HClO_4$-HNO_3 acid mixture (McKeague 1978). Lattice or mineral K contents were calculated from the difference between total K content of the sample and the boiling HNO_3- K. K contents in the extracts were measured using an ICP-OES.

Statistics
Soil physico-chemical properties, soil potassium in various forms and crop variables were subjected to analyses of variance (ANOVA) using a general linear model to simultaneously test the main and interactive effects of soil grouping (volcanic versus non volcanic), garden type (old versus new) in different soil depths (top and sub surface). Pearson's correlation coefficient was worked out between and/or among different forms of K. Genstat Discovery Edition was used for the statistical analysis.

Results
The effects of soil groupings (volcanic or non-volcanic) and garden type (old or new) on different K fractions are given in Table 1. Only main effects are presented as interactions between garden types, soil groupings and soil depths were non significant. Water soluble K content is the immediately plant available forms of K. It varied widely among soil groupings, ranging from 2.8 mg/kg to 38.6 mg/kg with significantly higher quantities occurring in non- volcanic soils. Water soluble K content constituted less than 1% of the total K. Exchangeable K content contributed to about 2.0- 3.1 % of the total K, while, non exchangeable K contents was in the range of 2.5- 4.2 %. Both, exchangeable K and non exchangeable K were significantly lower, in volcanic soils and old gardens, compared to non volcanic soils and new gardens (Table 1).

Exchangeable K which forms major pool of immediate plant- available K, was found to be significantly low ($P > 0.05$) in sweet potato gardens on volcanic soils and in older gardens. Potassium reserves (forms of K which are available slowly i.e., in medium to long- term) such as non exchangeable forms of K was found to be significantly low in volcanic soils and in old gardens (Table 1). Characterization of non exchangeable pools of K is quite important from the point of K nutrition in the long run. Non exchangeable K content was low in more than 95% of the top soil samples according to the criteria proposed for most of the field crops. According to categorization of non exchangeable K proposed by Srinivasarao et al (2007) soils with non exchangeable K contents less than 350 mg/kg are considered to be low in reserve K. Such soils should be managed with external application of potassium fertilizers. Old gardens showed lower non-exchangeable K contents than the new gardens despite the fact that they have greater contents of lattice and total K, indicating exhaustion of K reserves due to several cycles of sweet potato cultivation. In the new gardens which are due to go for planting, replenishment of K to non exchangeable pools could be expected. Due to the weathering action of soil minerals and release of K from residues and vegetation debris, K might be set free to soil solution which can inturn enter the inter lattice spaces. Non volcanic soils supporting sweet potato were in general rich in non exchangeable and lattice K reserves. Volcanic soils of PNG (Andosols) are reported to be dominated in hydroxy inter- layered vermiculite, imogolite/ allophane and volcanic glass (Bleeker and Sageman 1990; Rijkse and Trangmar 1995) while, non volcanic soils might have greater preponderance of illite, muscovite mica and K-feldspars (Sharpley 1989; Rubio and Gil- Sotres 1997; Rezapour et al.2009)

In the present study water soluble K content had no significant relation with exchangeable K and non exchangeable K (Table 2) which is in conformity to the some of the previous reports (Singh et al 1985; Sharma et al 2006). At any given point of time pool size of water soluble K is related to the pool size of exchangeable K. Exchangeable K which is the major form of plant available K, was highly significantly related to non exchangeable K and water soluble K in these soils. This indicates that there exists a dynamic equilibrium relationship between K in water soluble and K in inter-lattice spaces through exchangeable form of K. Depleted non exchangeable K could be replenished from the K in mineral fractions as indicated by strong positive correlation coefficient ($r= 0.728^{***}$), how ever, this is relatively time consuming, slow process depending on various soil and climatic factors.

Conclusion

The present study describes the status of different forms of K in sweet potato growing gardens of central highlands of Papua New Guinea. Although total K content of these soils are in acceptable range to that reported in other parts of the world, water soluble, exchangeable and non exchangeable forms of K are considerably low in older gardens. Especially, older gardens which are in volcanic soil groupings are more susceptible to the K depletion problem due to continuous sweet potato cultivation, possibly owing to their lower mineral K status. Such gardens should be managed either with allowing sufficient fallow period for regeneration of soil fertility or with suitable dose of mineral K fertilizers.

Table 1. Effect of soil groupings, garden types and soil depths on the distribution of K in different forms (mg/kg soil) in the sweet potato gardens of Central Highlands provinces of Papua New Guinea.

	Water soluble K	Exchangeable K	Non exchangeable K	Lattice K	Total K
Soil groupings					
Volcanic	10.51	63.80	75.57	2481.3	2625.7
Non-volcanic	14.41	97.81	189.1	5818.6	6101.7
F test significance	*	*	***	***	***
Garden type					
Old gardens	12.74	64.28	111.8	4166.1	4390.8
New gardens	12.18	97.33	152.8	4133.9	4336.5
F test significance	ns	*	*	ns	ns
Soil depth					
Top soil	12.13	95.50	132.0	4722.7	4955.0
Sub surface soil	12.80	66.13	132.7	3522.7	3772.4
F test significance	ns	ns	ns	ns	ns

Statistical significance of differences between mean values: $P > 0.05$, ns; * $P < 0.05$; *** $P < 0.001$

Table 2. Pearson's correlation coefficients (r) between /and among different forms of K.

	Exchangeable K	Non exchangeable K	Lattice K	Total K
Water soluble K	0.254^{*}	0.250*	0.242^{ns}	0.121^{ns}
Exchangeable K		0.324**	0.223^{ns}	0.242^{ns}
Non exchangeable K			0.728***	0.746***
Lattice K				0.999***

$P > 0.05$, ns; * $P < 0.05$; *** $P < 0.001$

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Distribution patterns of Collembola affected by extensive grazing in different vegetation types

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Abstract
The effect of extensive grazing on Collembola communities was studied in different vegetation types of National Park Neusiedler See – Seewinkel (Austria). The results showed that the effect of extensive grazing leads to differences considering soil temperature, bulk density and organic matter content. The total Collembola material included 6338 specimens belonging to 20 euedaphic and eight hemiedaphic species. Significantly higher values of total species richness and abundance of hemiedaphic species were recorded at grazed plots in comparison to ungrazed plots. However, Collembola species number was correlated positively with plant species richness and collembola density was correlated positively with organic matter content and soil temperature. The ratio of euedaphic to hemiedaphic species was negatively related to bulk density and positively to plant species richness. Canonical correspondence analysis based on collembolan communities belonging to different vegetation types suggests separate trends for grazed and ungrazed plots. The main environmental factors correlated with the changes in collembolan communities were plant species richness and soil bulk density.

Key Words
Microarthropods, life forms, land-use, grasslands.

Introduction
An effective way of land-use, extensive grazing by large herds of cattle in the National Park Neusiedler-See Seewinkel, eastern Austria, was studied. Unfortunately it disappeared for economic reasons by the 1960s and was re-established only in 1987 as a management practise for conservation of the remaining dry to sub-humid grasslands. Extensive grazing results in high structural heterogeneity by browsing, trampling and nutrient in- and outputs in the habitat (Fischer *et al*. 1996). Moreover, livestock grazing influences plant-community structure, soil temperature, moisture and soil compaction and is also likely to effect directly or indirectly the populations and diversity of soil biota (Bargett and Wardle 2003; Clapperton *et al*. 2002; Merill *et al.* 1994; Peterson *et al.* 2002). Collembola are among the most ecologically diversified microarthropods occurring in high abundances in grasslands. They play an important role in plant litter decomposition processes (Anderson *et al.* 1983; Faber *et al.* 1992) and in forming soil microstructure (Dunger 1983). The relatively large population sizes and potential influence on nutrient mobilization make suggests that they represent a key group for determining system's productivity.

In the present study, we examined the effects of extensive cattle grazing on collembolan communities in five different vegetation types. Moreover, we analysed how different ecological life forms (euedaphobionts and hemiedaphobionts) reacted to extensive grazing mirroring the severity of disturbance caused by grazing in different soil depths. The main questions of the present study were: How do euedaphic or hemiedaphic species respond to grazing? Which main habitat parameters influence the distribution of Collembola communities in grazed and ungrazed grasslands?

Methods
Site description
The investigation was performed in the National Park Neusiedler See – Seewinkel, about 40 km southeast of Vienna, Austria (45°46'N, 16°47`E).The study area represents sandy salt-effected chernosems with low nutrient status. The diversity, density and community structure of Collembola were investigated at two adjacent plots, one of which has been extensively grazed by cattle from May to October for 20 years (W-G). The investigated pasture was managed with low grazing intensity (<2 BCU/ha). The second plot was an ungrazed, densely vegetated exclosure (W-U).

Both grazed and ungrazed plots were characterized by five vegetation types of different plant species richness growing along the dry-to-wet gradient:

W1-species rich semi-dry grassland (*Carici stenophyllae- Festucetum pseudovinae)*,
W2-salt-effected and periodically flooded grassland (*Centaureo pannonici-Festucetum pseudovinae*),
W3-species poor reed area (*Scorzonero parviflorae-Juncetum gerardii, Scirpetum maritimi*),
W4-species poor *Agrostis* grassland (*Scorzonero parviflorae-Juncetum gerardii*, *Scripetum maritimi*),
W5-*Puccinellia* grassland (*Atropidetum peisonis*);

Collembola sampling
In April, July and September 2005, 10 soil cores of 10 cm^2 in soil surface and 10 cm deep were taken from each of the vegetation types of grazed and ungrazed sites. Collembola were extracted using Tullgren funnels, counted and determined to species level. To examine whether Collembola species in different soil depths are differently affected by extensive grazing Collembola were classified to hemiedaphic species (living in the litter and in upper layers of the humus horizon) and euedaphic species (particularly living within the soil) after Rusek (2007).

Data analysis
We used pair/sampled Wilcoxon-Tests to compare the grazed and the ungrazed sites regarding abiotic parameters (phosphate, nitrate, pH, organic matter content, soil temperature, bulk density and plant species richness) and Collembola parameters (species richness, total number of individuals). In order to assess the influence of abiotic parameters on Collembola parameters (species richness, total number of individuals, ratio euedaphic:hemiedaphic species and individuals, respectively) stepwise multiple regression analyses were conducted. Response variables were tested for normality with Shapiro-Wilk test and log/transformed when necessary to meet criteria for statistical analysis. The number of variables to enter a regression model was limited to two in order to avoid problems related to overfitting. We used an ordination method (Canonical Correspondence Analysis, CCA) to check for the influence of environmental factors on the species composition of the Collembola communities.

Results
Environmental variables
The effect of extensive grazing leads to significant differences considering soil temperature, bulk density and organic matter content. Soil temperature at the ungrazed plots was considerably higher by 3% than at the grazed plots ($P = 0.043$) Furthermore, bulk density was respectably higher by 8% at the grazed plots than at the ungrazed plots ($P = 0.043$). On the other hand, organic matter content at the grazed plots significantly exceeded by 24% tthat of the ungrazed plots ($P = 0.043$). Plant species richness was higher on grazed plots than on ungrazed plots but the difference was only marginally significant ($P = 0.66$).

Collembolan diversity
A total of 6338 specimens belonging to 20 euedaphic and eight hemiedaphic species of Collembola were obtained from the ungrazed and grazed plots. Total diversity was considerably higher at the grazed plots than at the ungrazed plots ($P = 0.039$).The total abundance of Collembola varied between 26,000 ind./m^2 on grazed plots and 16,200 ind./m^2 on ungrazed plots but differences were not significant. Considering the diversity of both collembolan life forms only hemiedaphic species reached remarkably higher densities at the grazed site ($P = 0.043$).

Total species richness of Collembola was increased by means of the number of plant species ($r^2 = 0.630$, $P = 0.006$). Collembola density was positively correlated with the amount of organic matter ($r^2 = 0.434$, $P = 0.038$) and soil temperature ($r^2 = 0.468$; $P = 0.029$). The ratio of euedaphic to hemiedaphic species was negatively related to bulk density ($r^2 = 0.609$, $P = 0.008$) and positively to plant species richness ($r^2 = 0.561$, $P = 0.013$); i.e. that the number of hemiedaphic species increased relative to the number of euedaphic species with increasing bulk density and decreased with increasing plant species richness. This shift was mainly caused by changes in euedaphic Collembola diversity: the number of species of this life form increased with plant species richness and soil organic matter ($r^2 = 0.882$, $P = 0.001$; plant species richness: partial $r^2 = 0.770$, soil organic matter: partial $r^2 = 0.112$) and declined with increasing bulk density ($r^2 = 0.590$, $P = 0.009$) while hemiedaphic species richness appears to be unaffected by bulk density and all other environmental variables (all P-values > 0.05).

The CCA analysis (Figure 1) revealed a clear differentiation between the Collembola communities of the different vegetation types and of grazed and ungrazed plots. The first canonical axis, which explains 32.2% of variation in the species data, was primarily determined by plant species richness but also by soil bulk density. Moreover, the latter was also correlated with the second canonical axis (19.9% explained variation). The model gained from forward selection containing both variables; plant species richness and soil bulk density, explained 53.1% of the variation in the Collembolan species data ($P = 0.001$).

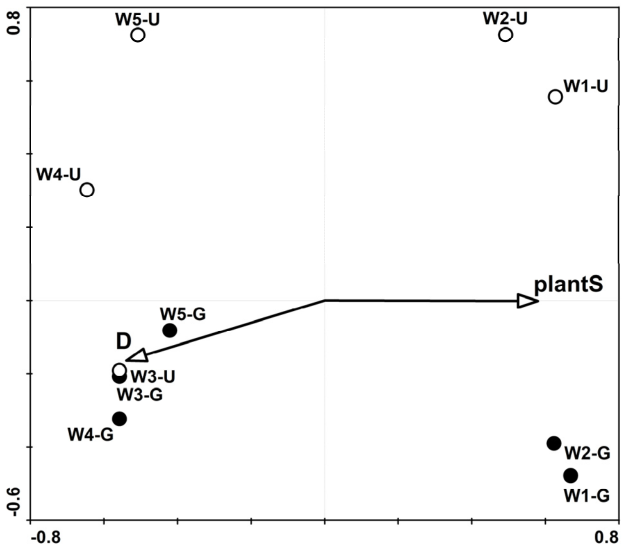

Figure 1. CCA biplot based on collembolan communities in different grazed (W1-G to W5-G) and ungrazed (W1-U to W5-U) vegetation types and environemental factors bulk density (D) and plant species richness (plantS).

Discussion

We found that extensive grazing had considerable influence on habitat parameters but also on collembolan communities. Grazing treatment significantly affected soil water content, soil temperature and bulk density. The results of a higher amount of organic matter at grazed plots could be attributed to the fact that livestock trampling can induce pressing of the dead plant and old grass material to the soil surface and therefore advancing a dense sward (Nitsche and Nitsche 1994). Species richness of Collembola was remarkably higher at grazed plots than at ungrazed plots. Extensive grazing represents a low-input system with increased habitat and resource heterogeneity resulting in a more diverse fauna (Morris 2000; Clapperton *et al*. 2002).

According to the present study significant correlation between Collembola abundance and grazing effect was found for hemiedaphic species, which developed respectably higher densities at grazed plots. We have found that Collembola density is positively correlated with organic matter content which agrees with other studies (Curry 1987; Eaton 2004). Most of the euedaphic species which reacted negatively to the higher bulk density are more sensitive to disturbance by trampling and compaction of the soil than hemiedaphic species are. Hemiedaphic Collembola seems to be more tolerant to drought, trampling and other disturbances caused by livestock grazing.

CCA results show that the collembolan community structure differs depending on the presence or absence of livestock grazing. The main environmental factors correlated with the changes in collembolan communities are the plant species diversity and bulk density.

Conclusions

Extensive grazing has considerable influence on habitat parameters and on distribution of Collembola as well. The effect of grazing induced increase in species richness of total Collembola. Separate trends in Collembola life forms were suggested for grazed and ungrazed plots. The number of plant species, the amount of organic matter,bulk density and soil temperature were the most important of the measured habitat parameters at grazed and ungrazed plots that correlated with density and diversity of Collembola. Extensive grazing as suitable land-use has a strong impact on Collembolan communities as well as Collembola life forms as a group.

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Effects of parent material and land use on soil phosphorus forms in Southern Belgium

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Abstract
Appropriate management of soil phosphorus fertility should rely upon sound knowledge about the phosphorus reserve and its bioavailability. However, the fate of P in the soil depends on soil characteristics and agronomic practices. The influence of parent material and land use on P content and distribution between organic and mineral pools has been studied. The diversity of pedological contexts in the Walloon Region reflects on soil properties. A large range of P contents has been observed and partially attributed to an effect of parent materials in surface as well as in deeper horizons. Relationships with total Al and Fe contents were observed. Differences among parent materials were observed for available P but not for proportions of organic and inorganic P. Land use had also an influence on P content but only for surface horizons. Pasture soils presented higher P contents than crop soils but lower available P. Other parameters, such as pH or organic matter which depend on land use, also seem to have an impact on P availability. This confirms that the management of phosphorus resources in cultivated soils has to take into account the sub-regional specificities of soil parent materials and land use.

Key Words
Total phosphorus, organic phosphorus, inorganic phosphorus, available phosphorus, fertility.

Introduction
Soil phosphorus faces both environmental and agronomic issues because it is responsible for eutrophication of surface waters and, at the same time, it is an essential element for plant growth. It is therefore important to deepen our knowledge about the quantities and forms of P in soil and hence about the susceptibility to migrate to other compartments of ecosystems. Reserves of P in soil can be divided into organic (P_{org}) and inorganic (P_{inorg}) fractions. The absolute and relative importances of these reserves vary according to environmental conditions. In low input farming systems, the reserves of organic and inorganic P may be a significant source of P to crops (Romanya and Rovira 2009). Actually, when negative balance of P occurs, the availability of P to crops is determined largely by weathering and/or mineralization of soil P reserves. These processes are influenced by soil biota and soil organic matter dynamics. Most P used by plants is taken up in inorganic forms but organic P reserves can represent a large proportion of total P content in soil. According to Fardeau and Conesa (1994), P_{org} represents 25-30% of total P in cultures and up to 75-80% in grasslands and forests. So, P availability to plants depends on (i) the chemical balance of P_{inorg} between the solid phase and soil solution, (ii) the microbial decomposition of plant litter and (iii) the pools of organic P in the soil (Tate and Salcedo 1988). Soil organic P reserves can therefore play a significant role in supplying P to plants. However, a large proportion of organic P reserves may not be readily available to plants (Romanya and Rovira 2009).

There are two main P sources in agricultural soils: the natural geochemical background and the fertilization. Soil forming processes should act on the P fractionation. According to some authors, the P_{org} contents should be higher within well developed soils and P_{inorg} should become less available because of the occlusion of P in minerals. The distribution of soil P between different organic and inorganic forms depends on agronomic practices and soil properties. Chardon and Schoumans (2007) have shown the effect of the soil texture on the P behaviour in soil-sediment systems. The organic matter, clay content or other bounding sites like Al and Fe oxides influence the forms of P. Fertilization practices have also an impact on the forms of P. Romanya and Rovira (2009) have shown that organic P reserves were sensitive to the quantity of P added, rather than to the quality of the input and that organic P reserves were high in the soil that received large amounts of manure.

The evaluation of soil P fractions is essential to determine the P behavior in soils. The objectives of this paper were (i) to evaluate the influence of parent material and land use in soils from Southern Belgium on P content, and (ii) to study the fractionation into organic and inorganic pools.

Materials and Methods

Study Area and Soil Sampling

In order to be representative of the diversity of natural environment in Southern Belgium, twelve types of parent materials were first selected according to their spatial importance and to differences in their properties. For each parental material, a representative area was chosen according to geology, pedology, relief and land use (Figure 1). Within each area, the one to two dominant soil type / land use combinations were selected. Ten parcels from 10 different farms were sampled for each parental material. Soil types were observed on the field after augering and soil samples were taken in surface horizon (0-15 cm) and in the deep 100-120 cm horizon. An intermediate horizon has been sampled in some occasions in order to evaluate P profiles. Finally, 258 samples were collected, 120 surface samples, 120 deep samples and 18 intermediate samples. All the samples have been taken between October and December 2008. They were located in 76 fields, 15 temporary grasslands and 29 pastures.

Chemical Characterization

Samples were dried at 40°C and sieved at 2 mm (an aliquote at 200 µm) prior to storage and laboratory analyses. Some soil characteristics were determined. Particle size distribution was determined by sedimentation according to pipette method. The cation exchange capacity (CEC) was determined by modified Metson method. Total organic carbon (TOC) content was analyzed by Springer-Klee method (ISO 14235, 1998) and pH $_{water}$ (v:v, 1:5) and $pH_{1N\ KCl}$ (v:v, 1:5) according to ISO 10390 (2005). Available phosphorus (P_{av}) was determined following Lakanen-Erviö method (Lakanen and Erviö 1971), and total phosphorus (P_{tot}) was determined after total solubilising by acid attack (NF X 31-147, 1996). Inorganic phosphorus (P_{inorg}) was extracted by addition of 20 ml of H_2SO_4 to 0.5 g of 0.2 mm soil. This mix was brought to the boil during 10 minutes and put in a 100 ml balloon. Then, the solution was filtered and P content was measured. Organic phosphorus (P_{org}) was determined by calculation: P_{tot} - P_{inorg}. After extraction, phosphorus contents were measured by colorimetry (Murphy and Riley 1962). The metals in all extracts (Fe, Al and Ca) were determined by flame atomic absorption spectrometry (VARIAN 220).

Statistical Analyses

The effects of land use and parental materials on soil P fractions were evaluated for each depth, with analysis of variance using the General Linear Model Procedure of MINITAB 15 program. These differences were considered significant at the $p \leq 0.05$ level. A correlation and regression study has also been made between the various parameters to identify significant relationships with phosphorus forms.

Results and Discussion

Soil properties

Large ranges of variation have been observed for studied soil properties. Soil pH_{water} ranged from 4.62 to 8.4, TOC from 0.42 to 9.71%, clay content from 6.2 to 67.2%, and CEC from 2.5 to 71.2 $cmol_c$/kg. These values illustrate the diversity of studied contexts which are representative of diversity within Southern Belgium.

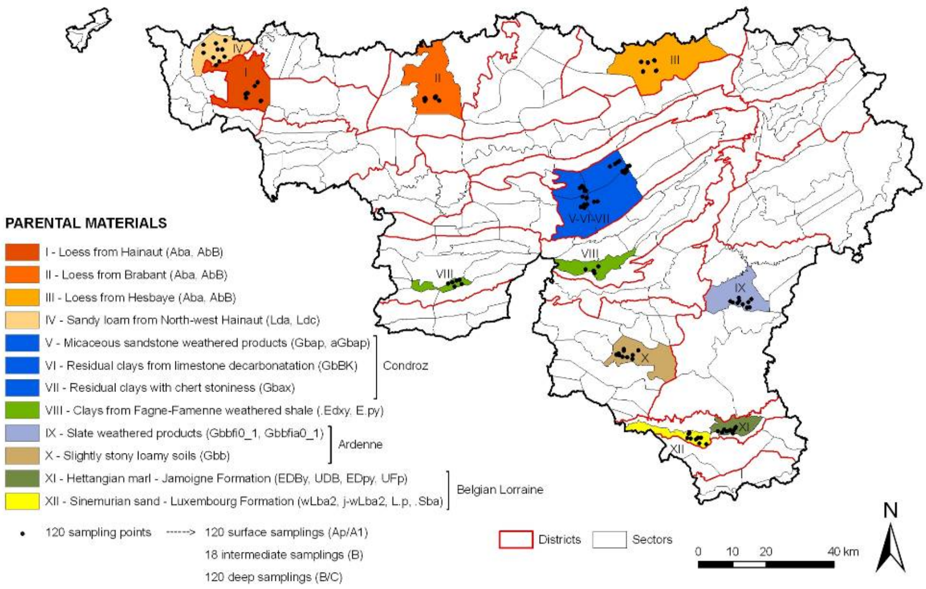


Figure 1. Location of studies samples in Walloon Region.

The soil P_{tot} values ranged from 167 to 2294 mg P/kg, with a mean in surface and deep horizons of 894 and 415 mg/kg, respectively. Leaching is considered as negligible in Southern Belgium, so deep P content can be considered as natural content, resulting from the weathering of soil parent material. The difference between surface and deep content can be attributed to effects of biogeochemical cycle and fertilization inputs. This hypothesis doesn't take into account the possible disturbation due to mesofaunal activity but the importance of these processes seems very difficult to evaluate. On average, deep soil P amounted to half of surface P. Available P represented on average about 9% of the total P in surface horizon. However, this percentage can vary highly according to soil properties.

Effects of parent materials

Parent materials (MP) had a significant influence on P forms and on P, Al, Fe and Ca contents in both surface and deep horizons. A South-North gradient has been observed for P_{av} in Walloon Region, which confirms previous results.

P_{tot} and P_{av} present opposite distributions among parent materials (Figure 2). The lower P_{tot} (MP I to IV), the higher P_{av} content, it could be explained by differences in P sorption capacity. This soil property is very important and varies from one soil type to another. Soil differences can generate different P behaviour. In light textural soils like sandy and silty soils, P is more readily available for plants and for lateral transfers. These areas are the most sensitive for regarding surface water eutrophication.

Effects of land use

Figure 4 illustrates the very highly significant differences of P status within the three studied land uses. In cultivated soils, P_{tot} represented 81% of that in permanent pastures. Differences in P input/output balances, with higher exportations in crop soils, could explain these differences of P status. Similar differences were also observed for other P forms, except P_{av}. The amounts of available P in the cultivated soils were in average 65% and 138% higher than permanent pasture and temporary grasslands respectively. These differences are probably linked to a higher availability of P brought by fertilization products in cultivated soils. Permanent pastures had higher P_{av} than temporary grasslands because manure was brought by animals during the pasturage. This leads to higher P content even if no significant differences can be shown. According to phosphorus forms, temporary grasslands presented similar properties than permanent pastures, except for P_{inorg} which had an intermediate behavior between the two other land uses. Moreover, fertilization practices can vary deeply between parcels according to type of culture and plant exportations. Thus the P_{av} variability in cultivated soils is very important.

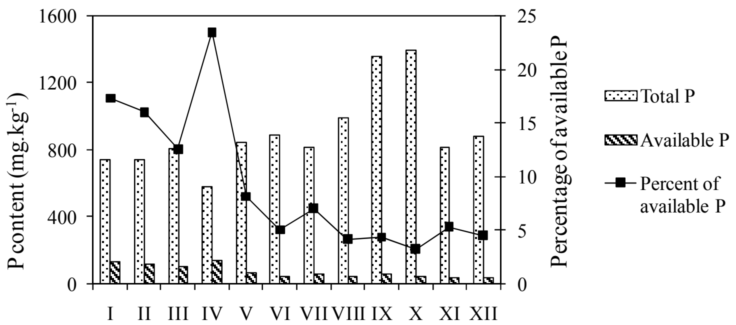


Figure 2. Effect of parental materials on total and available P contents in surface samples.

The P_{org} and P_{inorg} relative importance differed from one parent material to another but these differences were limited (Figure 3). On average, P_{org} represented 25% of P_{tot}, which corresponds to values given by Fardeau and Conesa (1994).

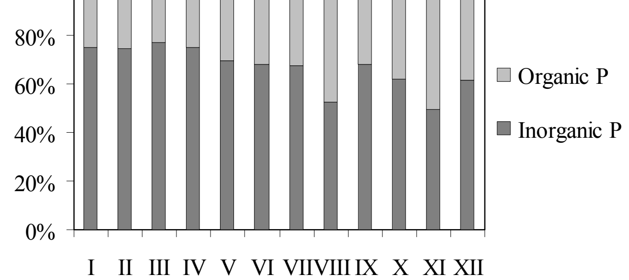
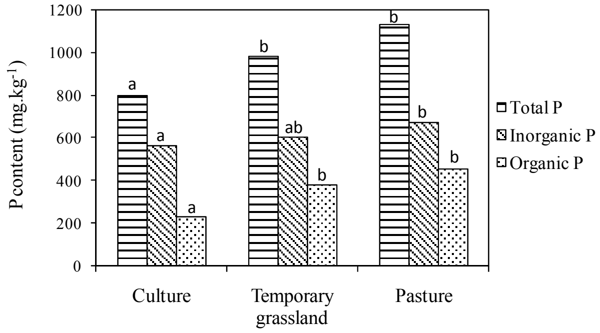
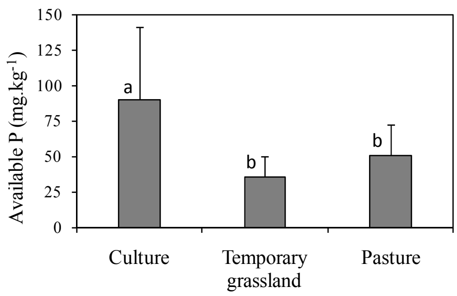


Figure 3. Organic and inorganic P fractions in surface samples grouped by parent material.



(a)



(b)

Figure 4. Effect of land uses on total, inorganic and organic P (a) and available P (b) for surface horizon.

Soil parameters influence

Soil properties had an important impact on P forms and P availability. Some parameters like total Fe or Al contents are considered as indicators of the quantities of adsorption sites for the phosphorus. Significant relationships were found with total P content. No correlation was found for clay or Ca contents.

Organic matter and pH were also related to P content. When levels of organic matter were high, so did P_{org} ($r=0.823^{***}$). However, levels of P_{av} were lower due to the P fixation on organic matter. For what concerns pH, the highest range coincided with highest P availability.

Conclusions

Parent material composition and land use are both responsible for differences in soil P contents. Parent materials influence the entire profile, while land use doesn't seem to influence deep horizons. Weathering is probably the most important process of P transformation in deep horizons. On average, P_{tot} in depth represented half of P in surface. Soils with the highest Al and Fe contents presented also the highest P_{tot} content but this P is not very available because of higher P sorption capacity. Some regions present higher environmental risks because of dominance of soils with low P sorption capacity and high P availability. This problem occurs in light textural soils under intensive agriculture. Indeed, crop soils have lower P contents than other soils but this P is highly available. So, the P management has to be thought according to the parental material and land use to avoid unsuitable effects for some regions.

Acknowledgements

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Impact of land use changes on soil carbon pools, gross nitrogen fluxes and nitrifying and denitrifying communities

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Abstract
The COSMOS-Flux project aimed at studying two situations that have important environmental impacts at a larger scale : the conversion tillage ⇆ no tillage where different tillage systems have been applied for 14 years at the start of experiment; the conversion grassland ⇆ annual crop where the introduction of temporary grassland into rotations is studied. The characterization of upper layers of soil for C and N pools, mineralization, immobilization and nitrification of N, along with characteristics of the nitrifying and denitrifying bacterial communities (activity, size and structure) were followed during 18 to 36 months after conversion. We observed that the tillage of soils untilled for 14 years, or the ploughing of the 5-year old grassland were major disturbances for the soils, which led to a very fast evolution of soil organic matter pools, N fluxes and microbial activities towards the characteristics observed for tilled and arable situations. Conversely, the shifts from till to no-till, and the establishment of grassland on soil previously cropped with annual species did not change significantly their soil characteristics at the time scale of the study. Among soil environmental variables, soil organic carbon appeared as a key driver of the observed responses.

Key Words
Environmental impact, land use, organic carbon, microbial function, tillage.

Introduction
Use and land management were long regarded as a problem to benefit primarily local, but now they appear as a factor with implications broader (Foley *et al.* 2005). Indeed, changes in uses and land management related to agriculture occurred and still occur over very large areas of the globe. These changes are very different: it may be conversion of forests into farmland or pasture, or crop intensification, for example. For over a decade and especially since the Rio Summit (2001), awareness of environmental issues related to climate change resulting from human activity, the role of vegetation cover and soil have significantly stepped up the questioning the impact of patterns of land use and agricultural practices. Management methods and their changes can have significant impacts on ecological processes and services provided by ecosystems such as the regulation of climate and biogeochemical cycles, the erosion control, soil formation, the role of 'habitat for living organisms, food production, biodiversity, etc. These practices and developments affect the ecosystem services either directly through their chemical or physical effects, or indirectly through their effects on biodiversity (composition change, number of species and / or alteration of the capacity of the species) (Diaz *et al.* 2007). In this context, microorganisms play a fundamental role in the functioning of the ecosystem including their role in biogeochemical cycles, they are key players in soils subjected to human influence.

The first objective of this project was to study how the characteristics of soil organic matters (quantity and quality), those of soil microbial communities (diversity, size, activity), and fluxes of C and N are coupled in the soil. The second objective of this project was to study how this coupling may be affected by changes in practice and effectively regulates the response of soils. Two situations that have important environmental impacts at a larger scale were investigated: the conversion tillage ⇆ no tillage and the conversion grassland ⇆ annual crop.

Materials and Methods
Experimental site
To study the conversion grassland ⇆ annual crop, the experimental site ORE ACBB (Agro-ecosystems, biogeochemical cycles and biodiversity) in Lusignan, France was chosen, which is studying the role of the introduction of leys in crop rotations (corn-wheat-barley). To study the conversion tillage ⇆ no tillage, the experimental site of Arvalis, Boigneville, France was chosen. On this site, a rotation pea-wheat-barley

received similar agricultural practices and the only differentiating factor for 14 years was the direct sowing (no tillage) and annual plowing. In both situations, the existing plots were split into two in order to follow the usual practices (controls), i.e. pasture aged 5 years and rotation of annual crops at Lusignan, direct sowing annual ploughing at Boigneville on half of the surface, while the other half underwent conversion: pasture ploughed or installed new pasture, ploughing of no tilled plots or abandonment of tillage. The choice to conduct parallel situations either in conversion or as reference treatment (control) (i.e. 4 treatments in parallel per experimental site), is essential to assess the evolution over time of several monitored parameters. Each site was sampled five times for 1.5 to 3 years, with more frequent sampling just after conversions.

Measurements on sampled soils
The characterization of upper layers of soil (0-20 cm and 0-30 cm for Boigneville and Lusignan, respectively) for C (total organic C, soluble C, biomass C and respiration) and N pools (total and mineral N), mineralization, immobilization and nitrification of N, along with characteristics of the nitrifying and denitrifying bacterial communities (activity, size and structure) were followed during 18 to 36 months after conversion. Soils from the 4 treatments were sampled at time 0, +3 weeks, +3 months, + 1 year, +2 years and + 3 years after the conversion, sieved. To assess the activity of nitrifying and denitrifying bacteria, we chose to measure the potential activity of nitrification and denitrification: these measures are often called Nitrifying Oxidizing Enzyme Activity (NOEA) (Smorczewski and Schmidt, 1991) and denitrifying Enzyme Activity (DEA) (Tiedje *et al*. 1989) in literature. This type of measurement is achieved by incubation of fresh soil samples to the laboratory in optimal conditions for achieving these bacterial activities in terms of temperature, oxygenation, and substrate availability. To determine gross N fluxes, we used the method of enrichment-dilution isotope ^{15}N. The gross mineralization is measured by enriching the pool of NH_4^+ with $^{15}NH_4{}^+$ and incubating during 24 hours at 15°C (Recous *et al*. 1999). Variations of isotopic abundance and quantities of ammonium nitrate and organic N were measured at the beginning and end of incubation. The rates of mineralization, immobilization and nitrification were calculated using FLUAZ software (Mary *et al*. 1998).

Results
The results first confirm other results that grassland, like direct sowing, install a large gradient of organic carbon in soils, one because residues crop are left on the surface of the soil where they decompose, another because the aerial litter and root of perennial grassland species accumulate in the first horizons below the surface soil. Apart from a possible increase in the total stock of organic C in soil (especially in grassland), a problem which was not the purpose of this project, the distribution of C, including the increased concentration in the “unperturbed” situations, causes a cascading series of "properties" for these layers of soil: increased amount of soluble carbon, increased heterotrophic microbial biomass and soil respiration, increased associated microbial transformations of nitrogen mineralization and immobilization, nitrification, denitrification . As many results on the two experimental sites were produced by this large project, results are presented in Figures 1 and 2 as examples.

The second important result concerns the effects of management changes. In the cases studied, the tillage of plots untilled during 14 years, and the ploughing of grassland aged 5 years, the common point is both the mechanical action exerted on the soil surface layer and the mixing of soil layers previously undisturbed, leading to the destruction of the gradient of organic matter and microbial communities and activities. In reality it is not possible to distinguish the two, however, (i) the first (mechanical) as a consequence of making "accessible" to the decomposition and mineralization of organic forms which were not: organic carbon " protected "by the structure of soil tillage on the one hand, root and aerial parts including roots at various stages of decomposition of the meadow destroyed, (ii) the second factor, the mixing of the layers, leads in terms of organic matter to immediate dilution of organic compartments, and microbial enzymes that cause medium-term adaptation of microbial communities to resources and new environmental conditions. The interpretations of the observed effects of these management changes, multi-factors in terms of the processes studied are complex and cannot actually access only a resultant of two effects.

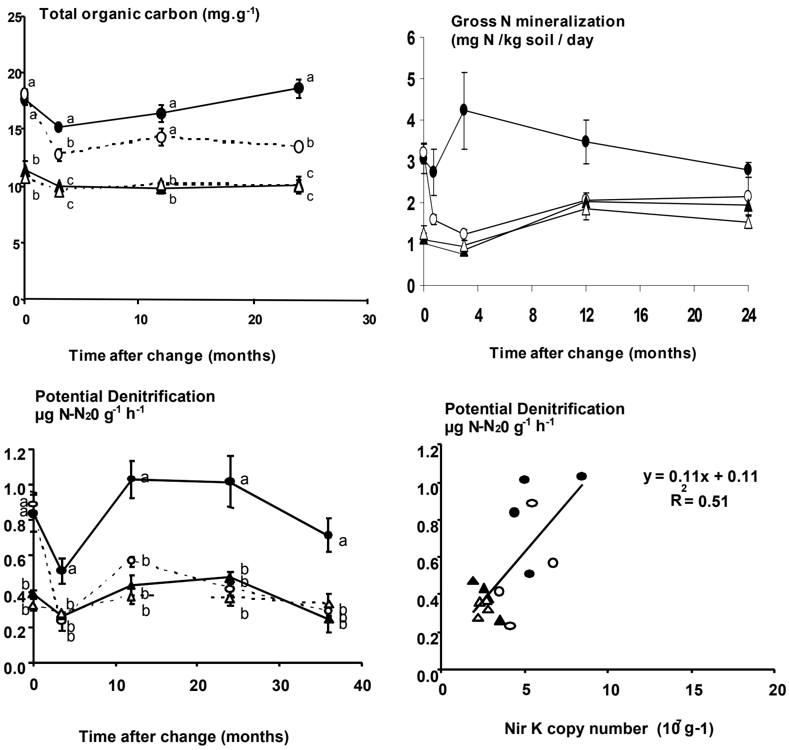

Figure 1. Changes in total organic carbon, the gross N mineralization, denitrification potential and relationship with denitrifier abundance (assessed by nir K copies) in grassland/arable crop conversion, at Lusignan site. ● continuous grassland, ○ploughed grassland, ▲ continuous annual crop rotation, △ establishment of grassland.

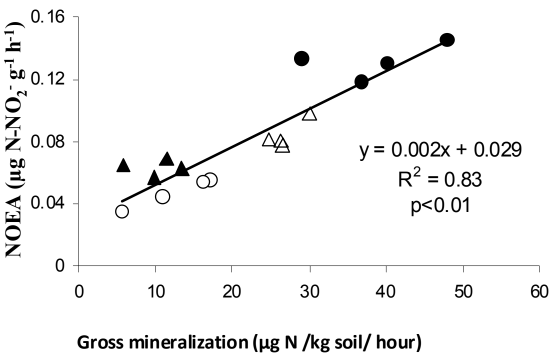

Figure 2. Correlation between nitrite oxidizer enzyme activity (NOEA) and gross N mineralization for soil samples taken from the layer 0-5cm, 2 weeks after the change of soil management on the Boigneville site. ● Continuous no till; ○ abandonment of no tillage; ▲ continuous annual ploughing; △ abandonment of annual ploughing.

Conclusions

The results of this project can draw several important conclusions about the effects of changes in use and management of soils but also transitions between these modes of management. This issue is important for example in the practice of direct sowing, which often includes periodical ploughing, or the establishment of agricultural scheme in conservation farming, or insertion of leys in rotations which aim to promote nutrient management based more on recycling organic materials. The results showed that in “reference” treatments, e.g. till vs. no-till soil, grassland vs. arable soil, soils had contrasted characteristics due to the significant gradient in the accumulation of organic C in the upper layer of no-till and grassland soils. We observed that the tillage of soils untilled for 14 years, or the ploughing of the 5-year old grassland were major disturbances for the soils, which led to a very fast evolution of soil organic matter pools, N fluxes and microbial activities towards the characteristics observed for tilled and arable situations. These effects result both from the cultivation of soil and the mixing of soil layers. Conversely, the shifts from till to no-till, and the establishment of grassland on soil previously cropped with annual species did not change significantly their soil characteristics at the time scale of the study, i.e. 18 months at Boigneville and 36 months at Lusignan sites, as these treatments appear as a cessation of disturbance rather than a disturbance. The results also provided the hierarchy of factors influencing the microbial communities involved in the nitrification and denitrification processes. The genetic structure of microbial communities hardly explained changes in their activity. Community size strongly explained changes in activity level for nitrite oxidizers, not denitrifiers. Among soil environmental variables, soil organic carbon appeared as a key driver of the observed responses.

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Impact of short rotation forestry on soil ecological services

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Abstract
Increasing prices for fossil fuels and enhanced efforts to reduce the CO_2 emissions induced a growing demand for renewable energy sources. Within this context woody biomass from short rotation forestry (SRF) on fallow ground or degraded land is an interesting option. Broad application of SRF may significantly influence - positively and negatively - soil ecological issues and services such as nutrient cycling, C-Sequestration, soil erosion, bio-remediation, biodiversity and ground water supply. Soil related results of the NOVALIS project, which aims to investigate a broad range of ecological effects of SRF at several sites in northern Germany since 2006 are presented. Results indicate that water consumption of SRF is relatively high and thus ground water recharge might be significantly reduced. Especially in more continental and dry areas or in year with less rainfall than 600 mm the ground water recharge might turn to zero. It is concluded that landscape specific planning and management strategies have to be applied to minimize water consumption of SRF. However, nutrient cycling especially for N is closed and efficient in SRF.

Key Words
Renewable energy, woody biomass, short rotation forestry, nutrient cycling, ground water recharge.

Introduction
Cultivation of Short Rotation Forestry (SRF) with poplars (Populus sp.) and willows (Salix sp.) for energy production is energy effective and coincides with several environmental objectives (Dimitriou *et al.* 2009a). Since an increase of cultivation of poplar and willow SRC has been projected in Europe, the consequent implications on water and nutrient issues and other ecological aspects like species diversity and landscape ecology have arisen (Dimitriou *et al.* 2009b; Baum *et al.* 2009). The given paper will focus on the aspect of water budgets and nutrient (N) cycling. Results reported were gained from the NOVALIS-project, started in Sept. 2006 and funded by the Deutsche Bundesstiftung Umwelt (DBU; Lamersdorf *et al.* 2008). The project aims to evaluate ecological benefits of SRF and encompasses aspects of soil ecology, silviculture, phyto- and zoodiversity, economy and landscape ecology. Several existing and newly created sites planted with clones of poplar and willows in north-eastern Germany were considered.

Methods
Water budgets for two sites with different climatic background conditions (more oceanic *versus* more continental) and development stages (older poplar *versus* younger willow) are described: 1.) the Georgenhof site, located in the uplands of Northwest-Hesse on weathered sandstone material with mean annual rain fall of 677 mm and a mean temperature fall of 7,8 ^{o}C. Various poplar clones were planted in 1995/96 on a former cropland, which was extended in 1992; 2.) the Cahnsdorf site, located in the Oberspreewald-Lausitz district on relatively poor and sandy periglacial deposit with mean annual rainfall of 563 mm and a mean temperature of 8,6 ^{o}C. Here, various willow clones were newly cultivated in 2007. Water flux simulations (model Hydrus-1D; Simunek *et al.* 1998) were applied for the period of 2006 to 2008 (Georgenhof), whereas 2006 was particular dry and 2007 more warm/moist. For Cahnsdorf a pre-plantation stage with a leave area index (LAI) below 1 was considered separately (2007) and was compared to a later stage of the plantation (2008, management phase, LAI of > 6).

Element budgets are presented for nitrogen (N) and the Georgenhof site. Given values are means of two investigated clones. Whole above ground tree harvesting as well as litter and soil analysis were applied in 2006/07. The buried bag method was applied to determine the internal N-flux in 2008. N_2O-emissions and NO_3 concentrations in the soil solution were only controlled during late autumn 2008. Thus annual flux rates were adjusted to pervious measurements in comparable SRF.

Results
The water budget for the Georgehof site (Figure 1) indicates strong losses of precipitation input by transpiration (50 %), interception (24 %) and evaporation (14 %). Only 10 % of the precipitation input contributes to the groundwater recharge. Mean values of the water budget varied significantly. In more dry years (2006 = 594 mm) almost no seepage output occurred, while in more wet years (2007 = 918 mm) 136 mm were transferred to the groundwater layer.

At the Cahnsdorf site and during the pre-plantation phase (Figure 2), 70 % of the precipitation input was transferred to the seepage output, while during the management phase, when the leave area index was set to 6 (Figure 3), only 40 mm, respectively 6 % of the input was seen in the seepage output. Like for the Georgenhof site the transpiration (52 %) and the interception (24 %) terms were the most prominent pathways for loosing water out of the system.

A positive N-budget with a surplus of 36 to 78 kg/ha/a was calculated for the Georgenhof site (Figure 4). The annual input of atmospheric N (ammonium and nitrate deposition) compensated most of the fixation of N by trees, respectively the output by harvesting the aboveground biomass (wood + bark, exclusive leaves). About 100 kg of N is internally circulating by leaf litter, up to 70 kg/ha/a are released by the net-nitrification process in the upper 30 cm of the soil. Thus the site is well supplied with nitrogen and there is no need for any N-fertilisation so far. According to all available data there is almost no emission of N_2O or loss of nitrate via soil solution.

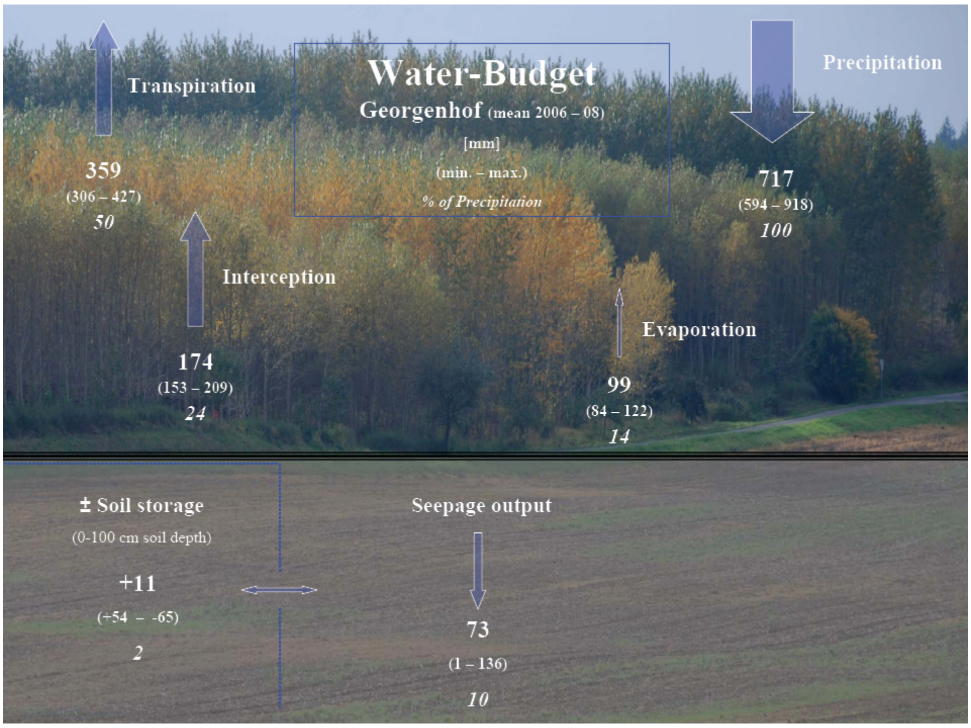

Figure 1. Water budget of the Georgenhof site (mean of 2006 to 2008).

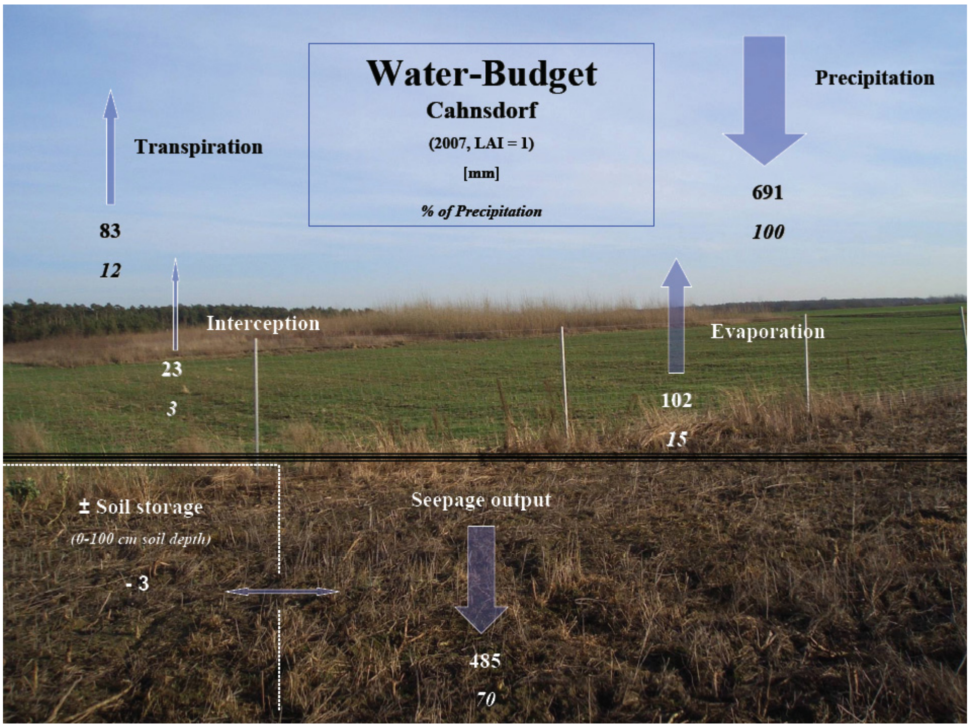

Figure 2. Water budget of the Cahnsdorf site, pre-plantations stage, i.e. the leaf area index (LAI) was set to 1.

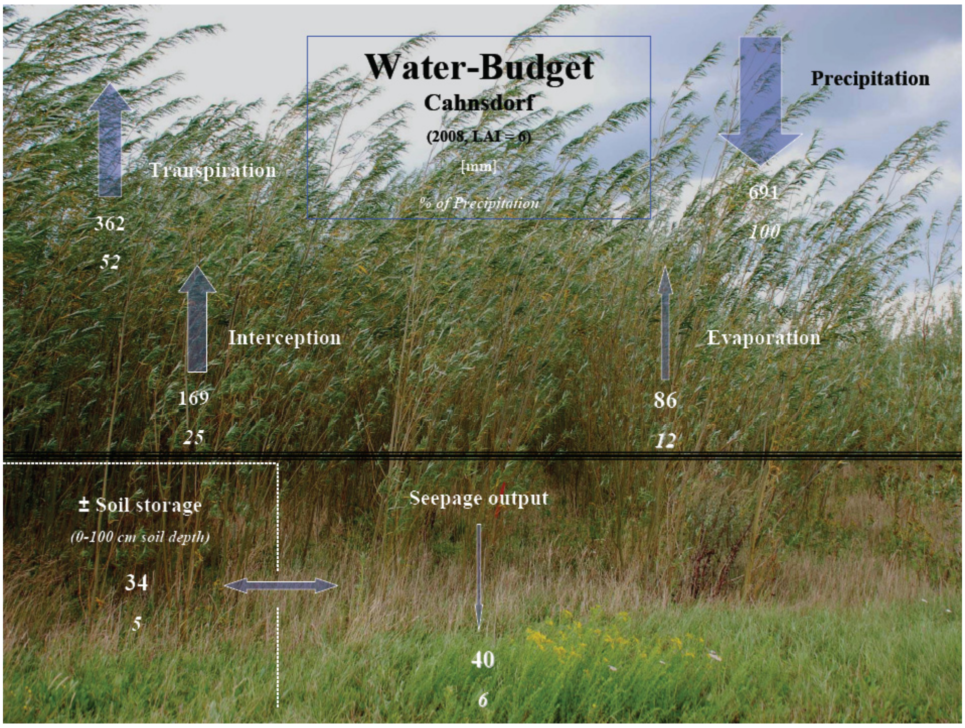

Figure 3. Water budget of the Cahnsdorf site, management phase, i.e. the leaf area index (LAI) was set to 6.

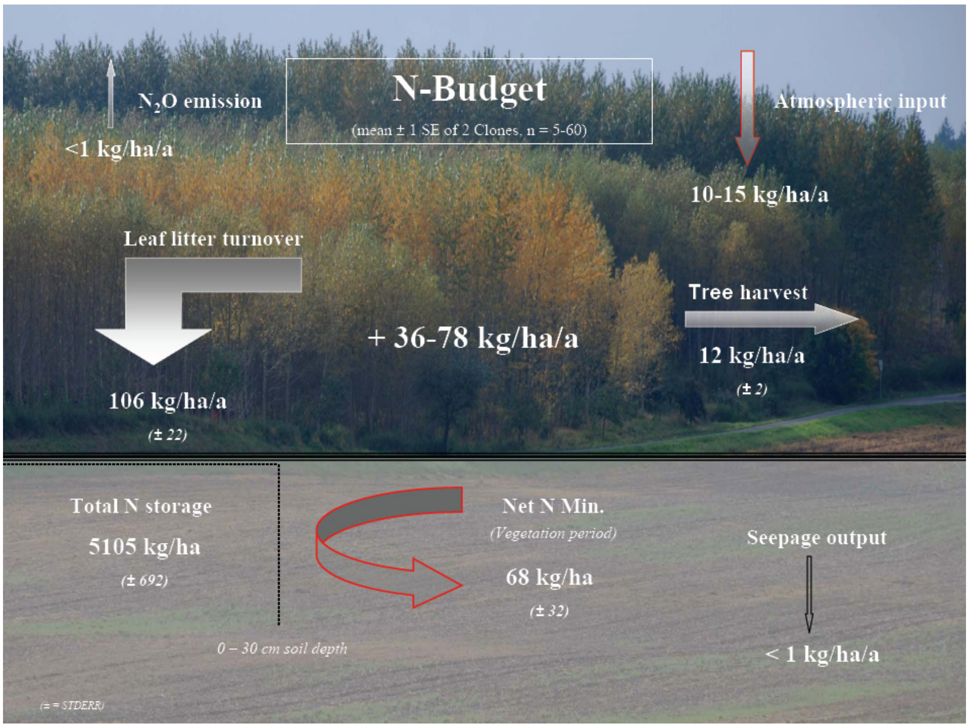

Figure 4. N-budget for the Georgenhof site.

Conclusions

Even under more favourable conditions (Georgehof 2007) the cultivation of SRF led to a significant reduction of the seepage and thus to an impaired ground water recharge. For relative dry years (2006) or more continental site conditions (Cahnsdorf site) results indicate an almost complete loss of seepage under SRF. It is claimed that before SRF will be installed the respective function of ground water recharge should be considered by a specific landscape planning and certain management strategies have to be applied to minimize the water consumption of SRF (e.g., reducing the rotation periods and minimizing edge effects to reduce the interception losses). The applied element budget for the Georgenhof site indicated a closed N cycling. According to all available data additionally mobilised (net-N-mineralisation, atmospheric input) is kept in the system and not transferred to the atmosphere (N_2O) or the ground water layer (NO_3). Furthermore there is obviously no need for any N-fertilisation so far.

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Impacts of conversion from forestry to pasture on soil physical properties of Vitrands (Pumice Soils) in central North Island, New Zealand

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Introduction

Tens of thousands of hectares of land have been converted from plantation forest to pasture in the central North Island of New Zealand between 2000 and 2010. The land use change was driven by the perceived better long term returns from dairy farming compared with forestry. Pumice Soils (NZ Soil Classification, equivalent to Vitrands in *Soil Taxonomy*) in the central North Island are formed on pumice deposited mainly from the AD 232 ± 5 Taupo volcanic eruption. The texture of Pumice Soils (Figure 1) varies from silt to coarse gravel and they have weak structure and erode easily when disturbed. Water holding capacity may be low but increases as the organic matter content of the topsoil is built up.

Figure 1. Upper part of soil profile on Pumice Soil, central North Island, New Zealand

When forests are cleared for pasture (Figure 2) the soil may undergo changes in soil structure affecting physical properties including the water infiltration rate and moisture holding capacity. Soil physical characteristics influence plant growth rates, soil erosion, and infiltration runoff, and therefore, flood occurrence in the catchment.

Soil organic matter (SOM) content is widely recognised as a factor that influences a number of soil physical properties (Bauer *et al.* 1992). De Oliviera *et al.* (2008) found that forest to pasture conversion caused lower organic carbon content (SOM). Steffens *et al.* (2008) similarly suggested that organic carbon, total N and total S concentrations decreased with increasing grazing intensity. However, recent findings from New Zealand showed a significant increase in soil organic matter in the first 5 years after conversion from plantation forest to dairy pasture on Pumice Soils (Hedley *et al.* 2009).

Figure 2. Land conversion from plantation forest (*Pinus radiata*) to dairy pasture near Taupo in the Central North Island of New Zealand

Pasture production on the Pumice Soils in summer is often limited by moisture availability. Because of increasing pressure on water resource use in the area an enhanced understanding of soil moisture holding capacity will contribute to our ability to manage plant available moisture. The plant root depth of much of the pasture on farms recently converted from forest was relatively shallow (about 10 cm), thus making pasture especially prone to moisture stress during dry periods. It is suggested that if we can identify causes of the shallow root depths and find a means to counteract the problem, root depth could be doubled from 10 to 20 cm, which would increase the moisture available to plants during dry periods and could conservatively give an overall 10% increase in pasture production.

The overall objective of this study was to investigate changes in soil physical properties in Pumice Soils following land use change from forest to pasture. Specific objectives were: (1) to investigate the consequences of conversion from forest to pasture on the soil moisture retention, plant rooting depth, aggregate stability, soil dry bulk density, and hydrophobicity; and (2) to determine the relationship between soil organic matter and the soil physical properties measured.

Site identification and experimental design

Four soil landscape units were identified: two near Tokoroa (Maxwell Farms) and two near Taupo (Wairakei Estate) (figure 3). On each soil landscape unit, seven sites were identified including current plantation pine forest, pasture established 1, 3, 4, and 8 years since forest-to-pasture conversion, long-term sheep pasture, and long-term dairy pasture.

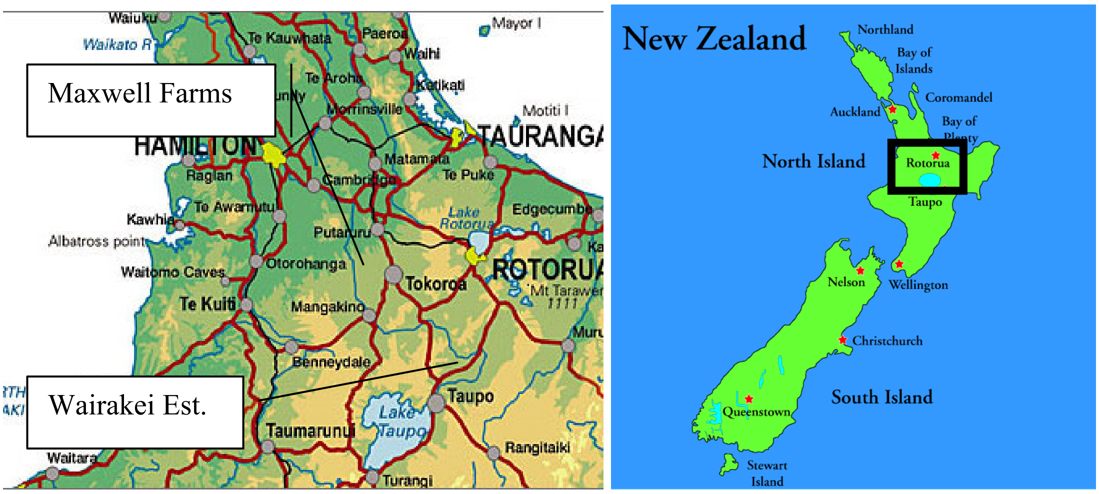

Figure 3. Location of study sites at Maxwell Farms and Wairakei, Central North Island New Zealand

Undisturbed soil cores were taken in triplicate from each site at a range of depths. Soils were analysed for unsaturated hydraulic conductivity, moisture release characteristics, readily and total-available water, particle density, and particle-size distribution. Soil water repellency or hydrophobicity was measured using the water drop penetration time. Total carbon and total nitrogen were determined in a parallel project (Lewis *et al*. 2010).

Discussion and conclusion

At the time of writing sampling is underway. Initial observations show that there are marked differences between forested, recently converted and long term pasture sites in terms of soil bulk density or soil structure and aggregation particularly in near surface soil horizons.

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Land use change in the tropics and its effect on soil fertility

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Abstract
Land use changes influence the fertility of the soil. Land use changes mostly focused on deforestation, cropland expansion, dryland degradation, urbanisation, pasture expansion and agricultural intensification. In tropical regions, forest is cleared for the expansion of cropland, wood extraction or infrastructure expansion. Croplands expanded by 50% during the 20th century, from roughly 1200 million ha in 1900 to 1800 million ha in 1990. There are several interacting drivers for land cover change but the exponential growth in human population is important. Currently, 95% of the population growth takes place in tropical regions and soil fertility in tropical regions is affected by rapid land use changes. The effects of deforestation and grassland conversions as well as agricultural intensification have been fairly well-documented but the spatial and temporal effects of soil fertility change and its interaction with land use change remains to be investigated.

Key Words
Land use change, tropical regions, soil fertility, deforestation, cropland.

Introduction
During the 20th century, the world population more than doubled from about 1.5 billion people in 1900 to 5.2 billion in 1990. Currently, the world population is growing by 1.3% per year compared to 2.0% growth in the late 1960s. More than 90% of the population growth takes place in tropical regions. About 80% of the population lives in developing regions; Asia accounts for 61% of the world total. The rate of population growth is declining and population will reach around 8.9 billion in 2050 (Lutz *et al.* 2001).

Despite enormous advances in remote sensing and GIS technologies in the past decades, systematic examination of trends in terrestrial land cover is yet to be made (Lepers *et al.* 2005). Most analyses of land-cover changes are based on data from remote sensing, censuses (statistical inventory, national, regional), and expert opinion through formal procedures. In order to integrate these heterogeneous data sources there is a need to determine the interrelationships between the data types (Lepers *et al.* 2005). There is a need to understand land cover changes and its effect on the overall ecosystems. Land use change affects the global climate via the carbon cycle, the water cycle through changing evapotranspiration and hydrological regimes but land use change also affects biotic diversity, soil degradation, and the ability of biological systems to support human needs. In other words, such changes influences earth system functioning (Lambin *et al.* 2003).

We roughly know where land use changes occur and we also know that land use changes affect soil chemical and physical properties. Such changes have been fairly well-documented but a systematic global scale link between land cover change and soil fertility change has - to our knowledge - not been made. Here I review the major patterns in land use and land cover change in the tropical regions and how these changes affect soil fertility and nutrient management.

Land use change
Lambin *et al*. (2003) used five categories of land cover change: cropland, agricultural intensification, tropical deforestation, pasture expansion, and urbanisation. Throughout this review, I shall more or less follow these categories to discuss trends in land cover change and its implications for soil fertility and its management. The area of cropland has increased from an estimated 300-400 million ha in 1700, to 1500-1800 million ha in 1990. The area under pasture increased from 500 million ha in 1700 to 3100 million ha in 1990. These increases led to the clearing of forests and the transformation of natural grasslands, steppes, and savannas. Forest area decreased from 5000-6200 million ha in 1700 to 4300-5300 million ha in 1990. The area under steppes, savannas and grasslands declined from around 3200 million ha in 1700 to 1800-2700 million ha in 1990 (Lambin *et al.* 2003).

The area under croplands increased by 50% in the 20th century from 1200 million ha in 1900 to 1800 million ha in 1990. This net increase in cropland area includes the abandonment of 222 million ha of cropland since 1900. There has been greater expansion of cropland areas since World War II than in the 18th and early 19th centuries combined. Significant changes in cropland occurred in southeast Brazil. Cropland expansion slowed down in the Midwestern USA, while there was abandonment in the eastern part. Cropland areas in northern Europe, the former Soviet Union, and China stabilized and even decreased in some regions, while it intensified in northeast China. Some croplands were abandoned in Japan. Clearing for cultivation continued in Southeast Asia and Oceania (Ramankutty *et al.* 2002).

Lepers *et al*. (2005) synthesised information on rapid land-cover change for the period 1981-2000 as part of the Millennium Ecosystem Assessment. They produced a series of global maps (10 by 10 km grid) that show how land cover has changed in the past decades. Some parts of the world were covered by several data sets, whereas for others only national statistics were available. As a result, some areas appear to be more affected by rapid land-cover change because they are studied more intensively. A summary of their findings: in Asia there are many areas where land-cover changes occur most rapidly; the Amazon basin is a hotspot of tropical deforestation and it mostly takes place at the edge of large forest areas and along major transportation networks. Deforestation occurs when forest is converted to another land cover or when tree canopy is reduced to less than 10%. Globally about 5.8 million ha is deforested each year, whereas annual forest regrowth is estimated to be 1 million ha. In tropical regions, forest is cleared for the expansion of cropland, wood extraction or infrastructure expansion.

Aerial photographs and satellite images

Several studies on land use changes used aerial photographs or satellite images from different periods combined with Geographic Information Systems (GIS). Holmgren *et al*. (1994) surveyed woody biomass on farmland in Kenya using aerial photographs and field measurements. A rapid increase of planted woody biomass was observed between 1986 and 1992 and the annual increase was estimated to be almost 5%. Population density was positively correlated with the volume of planted woody biomass: more people, more trees. The results imply that some pessimistic opinions on land-use development in Kenya are incorrect (Holmgren *et al.* 1994) and confirm some of the observations by Tiffen *et al.* (1994).

A study in Tanzania using normalized difference vegetation index (NDVI) imagery showed that the overall greenness increased between 1982 and 1994 (Pelkey *et al.* 2000). A detailed study in the Usambara Mountains in Tanga Region, Tanzania, showed a drastic reduction in forests cover from 53,000 ha in 1965 to 30,000 ha in 1991. Several studies from Kenya arrived at the same conclusions. In Embu region, Imbernon (1999) studied change in land-use in semi-arid and humid areas of Mount Kenya; tree cover decreased from 26% in 1956 to 24% in 1995. In the highlands North of Nairobi, Ovuka (2000) observed that in 1960 there was 15% fallow land but this had decreased to 6% in 1996. Woodlots had increased from 1 to 3% and coffee gardens from 0.2 to 12% over the same period. Areas without soil and water conservation practises increased from about 25% in 1960 to 70% in 1996. Most farmers depended on income from the land and thought that livelihood was better in 1996 than in 1960 (Ovuka 2000).

A study in Lake Malawi National Park, Malawi, using aerial photographs showed conversion of closed *miombo* to sparse woodlands (Abbot and Homewood 1999). Between 1982 and 1990, closed canopy woodland decreased by 7% whereas sparse woodland increased by 342%. In south-western Burkina Faso, Gray (1999) showed that between 1981 and 1993 the area under cultivation roughly doubled at the expense of scrub savannah. Human population doubled between 1971 and 1985. Tekle and Hedlund (2000) compared aerial photographs from 1958 and 1986 in the highlands of Kalu District, Ethiopia. A decrease in coverage by shrublands, riverine vegetation and forests was observed. Areas under cultivation remained more or less unchanged. It was concluded that land cover changes were the result of clearing of vegetation for fuel wood and grazing. In Papua New Guinea Ningal *et al*. (2008) assessed land use change in the Morobe Province (3.4 million ha) using topographic maps and Landsat^{TM} images. Between 1975 and 2000, agricultural land use increased by 58% and population grew by 98%; most new agricultural land was taken from primary forest. The forest area was more than halved from 9.8 ha per person in 1975 to 4.4 ha per person in 2000. Correlation between total population change and total land use change was strongly positive (64%).

Land cover change and soil fertility

The majority of land use changes are related to agricultural use of the land, including pastures. Agricultural activities change the soil chemical, physical or biological properties. Such activities include cultivation (mechanised, by hand), tillage, weeding, terracing, subsoiling, deep ploughing, manure, compost and fertiliser applications, liming, draining, irrigation and empoldering (Bridges and de Bakker 1997) but also biocides applications on cultivated crops may affect soil properties. Many soils have been improved since people started cultivation and soil improvements continue in many agricultural areas. Inputs are applied when needed by the crops, losses are minimised and environmental awareness and legislation have created agricultural practises that are ecologically and economically more sustainable. All these improvements are usually not reported in scientific literature. We do not have maps showing the great improvements in conditions in the past 100 years – quite the opposite: there is fair a body of literature on soil degradation in relation to agriculture.

Most of the concerns about soil degradation are fully justifiable, but hard data on the severity, extent, and impact are little which makes soil degradation a debated issue – particular in tropical regions (Hartemink 2006). A major factor in soil degradation is the soil chemical fertility and then in particular its decline as a result of the lack of nutrient inputs. This has been a major concern since sedentary agriculture started and is the main reason why farmers clear more land when farming in forested areas: the soil is depleted from plant nutrients.

The effects of land cover changes as well as subtle conversions on soil fertility are fairly well-documented. Land use change always affects soil quality and productivity. On-site effects are mostly related to changes in soil organic matter content. The most dramatic changes occur directly after a major land use conversion, such as deforestation. Mineralisation increases while the change in cover usually induces erosion and other landscape processes. Conversion has a large short term (1-5 years) on-site impact on soil properties such as soil organic C and bulk density, whereas land use intensification has longer term (10 to 80 years) effects on soil properties. We reviewed some studies in which the trends and rates of soil fertility changes were varying. Obviously, soil fertility is a complex issue consisting of several attributes that interact over time. Measurements require long-term research commitment as well as detailed knowledge about spatial and temporal variability. Most studies about the interaction between land use and soil fertility are on the profile and field scales which makes a direct link with spatial data on land cover change complicated. Systematic, consistent measurements of soil properties should be undertaken at a global scale, at a relatively fine resolution, since soil attributes are an important component of land cover (Lepers *et al.* 2005).

The ability of a region of the world to produce or access food is determined access to an adequate amount of productive cropland; the ability to maintain high crop yields on that land (often with the aid of external inputs, such as fertilizers, pesticides, and irrigation); or the ability to purchase and import food from other regions (Ramankutty *et al.* 2002). The majority of the world's fertile soils are already under cultivation. Much of the remaining cultivable land lies in marginal areas or in the richly forested regions of tropical Latin America and Africa. Clearing for cultivation implies a loss of forest. Prime farmland is lost to urbanization and the impact of soil degradation will further increase the pressure on the remaining croplands.

Conclusions

Globally, land use has changed considerably in the past decades – mostly reflecting the enormous growth in human population and their need for food. The world's population has doubled since 1960. The developing world accounts for about 95% of the population growth with Africa as the world's fastest growing region. The growing population has many implications but most of all it requires an increase in agricultural production to meet food demand. This demand can be met by expansion of agricultural land or by intensification of existing systems. Conservation and improvement of the natural resources on which agricultural production depends is essential. Soil degradation and in particular the decline of soil chemical fertility, is major concern in relation to food production and the sustainable management of land resources. It also affects land use but the spatial and temporal effects of soil fertility change and its interaction with land cover change remains to be investigated. Many studies on soil land use and land cover change are local and mainly aimed at specific systems such as shifting cultivation. More emphasis should be put on more intensive land use systems at more aggregated scales in a spatially explicit way.

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Land-use change from indigenous management to cattle grazing initiates the gullying of alluvial soils in northern Australia

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Abstract

Catchments in northern Australian have undergone dramatic land-use changes from traditional Aboriginal management to widespread cattle grazing post-European settlement. Quantifying the soil erosion impacts of these changes is essential to the future sustainable management of the soils in these catchments. Rates of gully erosion in alluvial soils were measured using recent GPS surveys and historical air photographs. The results indicated that median erosion rates currently and historically are within the same order of magnitude (0.1 to 1m per year). Historic air photo analysis demonstrated rapid increases in gully area of 2 to 10 times their initial 1949 area. Extrapolation of gully area growth trends backward in time suggested that most gullies initiated between 1880 and 1950. European cattle were introduced into the lower Mitchell catchment in the 1880's, suggesting the contribution of land use intensification to either gully initiation or acceleration. It is hypothesized that intense cattle grazing concentrated in the riparian zones during the dry season decreased perennial vegetation cover along hollows and steep river banks, increasing the potential for gully erosion. Once initiated on steep banks into dispersible sub-soils, alluvial gullies can rapidly progress in consuming and degrading the most productive part of the landscape, the riparian zone.

Key Words

Alluvial gully erosion, dispersible soils, cattle grazing, sediment dating, air photographs, historical analysis.

Introduction

Detailed remote sensing mapping within the 31,000 km^2 Mitchell River fluvial megafan has identified that active gullying into alluvial soils occupies a minimum of 0.4% (129 km^2) of the lower Mitchell catchment, with an estimated active front length of around 5,560 km (Brooks *et al.* 2009). It is estimated that these gullies erode more than 5 million tonnes of alluvial soil per year (Brooks *et al.* 2008). This soil erosion is concentrated along the riparian zone of the lower Mitchell River, where duplex soils that are strongly alkaline at depth have evolved from the original deposits of alluvial sand, silt and clay (BRS 1991). The erosion of these soils presents a major threat to both the local pastoral industry and downstream aquatic ecosystems. For example, the riparian 'frontage' of the Mitchell River is the most productive cattle grazing county in the catchment due to higher nutrient levels in younger soils as well as year round access to water. But it is also these dispersible riparian soils that are susceptible to accelerated erosion via gullying, which in turn pollutes downstream ecosystems with excess sediment. Therefore a management conundrum exists over the unsustainability of grazing these fragile alluvial soils.

Before soil conservation practices are implemented across catchments in northern Australia to reduce alluvial gully erosion of riparian soils, it is essential to investigate rates of alluvial gully erosion pre- and post-European settlement in order to better understand the potential human contribution(s) to land degradation. Data on erosion rates and initiation timing can be used in conjunction with process based studies on the driving and resisting factors for alluvial gully erosion, to more directly target remediation measures that will actually address the causes of human accelerated soil erosion. Some of the driving (climate, relief, floodplain hydrology) and resisting (soil chemistry, vegetation cover) factors for alluvial gully erosion are reviewed in Brooks *et al*. (2009). In this paper, preliminary results of historical alluvial gully erosion rates in the lower Mitchell catchment are highlighted, which support the hypothesis that the introduction of cattle and the concentrated grazing in riparian zones decreased vegetation cover along hollows and steep banks of the rivers, which subsequently promoted gullying and pushed the landscape across a threshold towards instability.

Methods

Recent erosion rates (2005-2009) were measured at 20 alluvial gully fronts (scarps) sites totalling 50,040m of common gully front repeatedly measured using *in-situ* differential GPS with sub-meter accuracy (Trimble with Omnistar High Precision). Accuracy depended on signal strength and vegetation cover, but was typically within 0.5 meters for repeat surveys. Historic erosion rates (1949 to 2007) were measured from decadal historical air photographs at 15 of the 20 study sites mentioned above. Digital copies of the photographs were georeferenced in ARC-GIS and the gully front locations were digitized, totalling 43,163m of common gully front repeatedly measured. Annual average linear erosion rates (m/yr) from GPS surveys and historic air photos were calculated by dividing the area change (m^2) over two consecutive time periods (years) by the scarp perimeter (m) measured at the later timer period. Trends in gully area over time were analysed using negative exponential functions well established in fluvial geomorphology (e.g., Graf 1977; Simon and Rinaldi 2006), but slightly modified for analysing changes in gully planform area over time.

$$\frac{A}{A_0} = a + be^{(-kt)} \tag{1}$$

where A is the exposed gully area at time t, A_0 is the initial gully area at $t_0 = 0$, a and b are dimensionless coefficients determined by regression, k is a coefficient determined by regression that defines the rate of change in gully area over time, and t is the time (years) since the initial starting point or the first air photograph.

As an indicator of land-use change over time, total cattle numbers on the land aggregation unit of Wrotham Park Station in the heart of the lower Mitchell catchment from 1880 to 2007 were obtained from the Queensland State Archives of historic lease applications. These data were also compared to data on total cattle numbers in Queensland from the Australian Bureau of Statistics. Insights into land condition prior to the introduction of cattle grazing, were gained from the diaries of Ludwig Leichhardt and John Gilbert who travelled down the Mitchell River together in June 1845, past numerous sites of currently extensive alluvial gully erosion (Gilbert 1845; Leichhardt 1847).

Results and Discussion

The median annual rate of scarp retreat was estimated from recent GPS measurements (2005-2009) to be 0.23 m per year across 50,040m of gully front. Annual rates calculated from historic photos (1949-present) at most of the same locations were within the same order of magnitude (0.1 to 1.0 m/yr), but with a higher median value of 0.37 m per year across 43,163m of gully front. Historic air photo analysis demonstrated rapid increases in relative gully area (A/A_0), with area increases 2 to 10 times the initial 1949 area (Figure 1). Values of the coefficient k in Equation 1 were small (0.0007 to 0.0028), indicating that erosion rates were near linear over time. However, rapid changes in erosion rates are often measured early in gully erosion cycles (e.g., Graf 1977), suggesting that the absence of pre-1949 photos might skew the trend lines toward a more linear form. Nevertheless, extrapolation of gully area growth trends backward in time, using Equation 1 and existing data, suggested that most gullies initiated (zero area) between 1880 and 1950. This is corroborated by tree ring analysis and OSL dating of gully age, results that are not presented here. Several gullies that were slow growing, with relative gully area (A/A_0) increases of less than 2 (Figure 1), had indeterminate initiation dates. However, these gullies had the largest k-values and strongest non-linear trends, suggesting that their erosion rates were higher early in the erosion cycle and trended towards an initiation point near European settlement in the late 1800's.

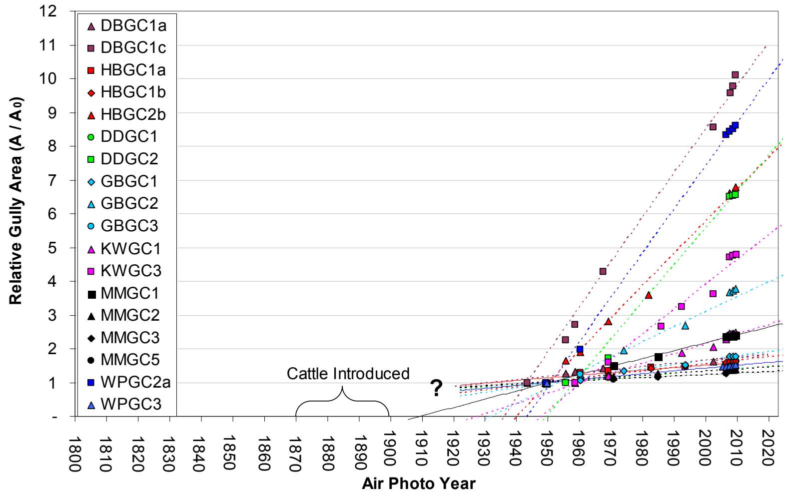

Figure 1. Relative changes in gully area over time, which is the ratio of the area at any time (A) and the initial gully area (A_0).

Historical data on the numbers of cattle in both the lower Mitchell catchment (Wrotham Park Aggregation) and the state of Queensland are shown in Figure 2. Cattle introduction in the Mitchell occurred around 1880, much later than in the rest of Queensland. The initial peak of cattle in the Mitchell catchment occurred between 1910 and 1920, similar in timing to the second Queensland peak. Subsequent fluctuations in cattle numbers both locally and regionally were strongly influenced by word cattle prices, droughts and grass availability, and the increased use of Brahman cattle (*Bos indicus*) in the early 1970's.

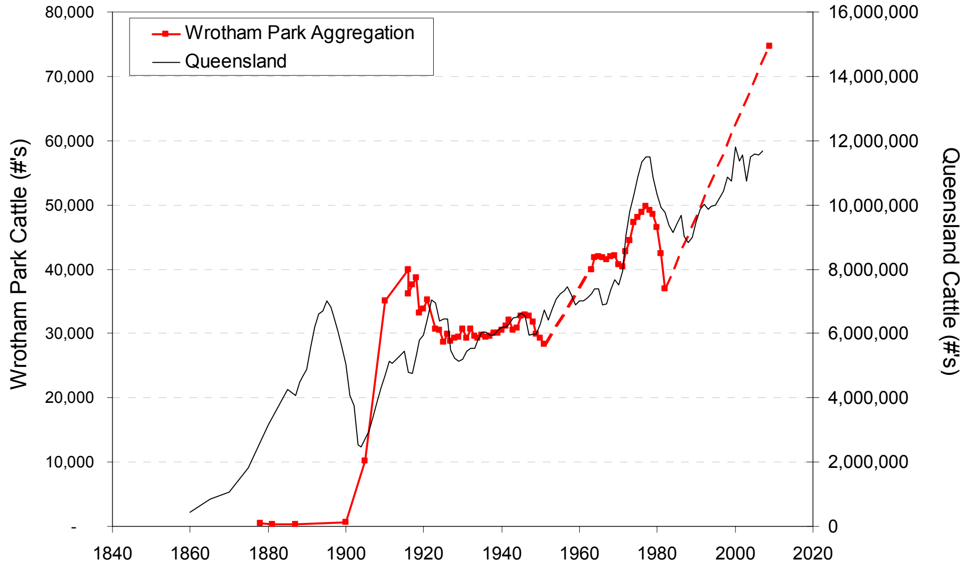

Figure 2. Historic trends in the numbers of beef grazing cattle at the historic Wrotham Park Aggregation cattle station (including Wrotham Park, Gamboola, Gamboola South, Highbury, Drumduff), in addition to historic trends in Queensland beef cattle numbers.

While the upward trend in cattle numbers (Figure 2) follows the upward growth trend in gully area (Figure 1), this correlation does not support a *continued* cause and effect relationship between cattle grazing and gully area. More likely, the similarity in these trends suggests that the *timing* of the initiation of many of these gullies coincided with the *timing* of the introduction of European cattle near the turn of the century. That is, the causal mechanisms for the initiation of these gullies might be different that the mechanism that continue to propagate these gullies across the landscape once initiated (Brooks *et al.* 2009).

Descriptions of the lower Mitchell landscape by Gilbert (1845) and Leichardt (1847) included references such as:

1. "The banks of the river were so steep, that the access to its water was difficult." (Leichardt 1847)
2. "[We] keep well back [from the river] to avoid the deep gullies frequent on the immediate banks of the river." (Gilbert 1845)
3. "….the [floodplain] was interrupted by gullies, and occasionally by deep creeks, which [were] the outlets of the waters collecting on the [floodplain]." (Leichardt 1847).
4. "We as usual had very distant gullies and hollows to cross; the banks of the river being very steep with very indifferent camping places." (Gilbert 1845)

These observations indicate that at least hollows and some form of gullies existed along the immediate banks of the Mitchell River pre-European settlement. However, nowhere in either Liechhardt (1847) or Gilbert (1845) were the terms erosion, eroded, bare, stripped, de-vegetated, dissected, unstable, incision, head cut, scarp, drop off, break-away, badland, or wasteland used to describe the soil surface, gullies, hollows, or creeks. These are terms and descriptions that would be used to describe these areas today (Brooks *et al.* 2009). It is more likely that they observed the precursor unchannelled hollows and small creeks that subsequently eroded into the massive alluvial gullies observable in air photos and measured in Figure 1.

Brooks *et al.* (2009) developed a conceptual model for the evolution of alluvial gullies in the Mitchell catchment over time. These alluvial gullies initiate as surface erosion and/or disturbance along unchannelled hollows and/or the steep banks of rivers and lagoons (Figure 3, stage 1a). As gully erosion proceeds and incises (cuts) into the dispersible sub-soils of the riparian zone, the dominant sources of water for erosion switch from surface floodwater and rainfall runoff to subsurface seepage erosion and erosion by soil piping. Once initiated, the data from Figure 1 suggests that erosion continues at a relatively consistent rate regardless of driving factors such as climate or changes in vegetation cover from grazing or fire. These factors are likely more important in the gully *initiation* process.

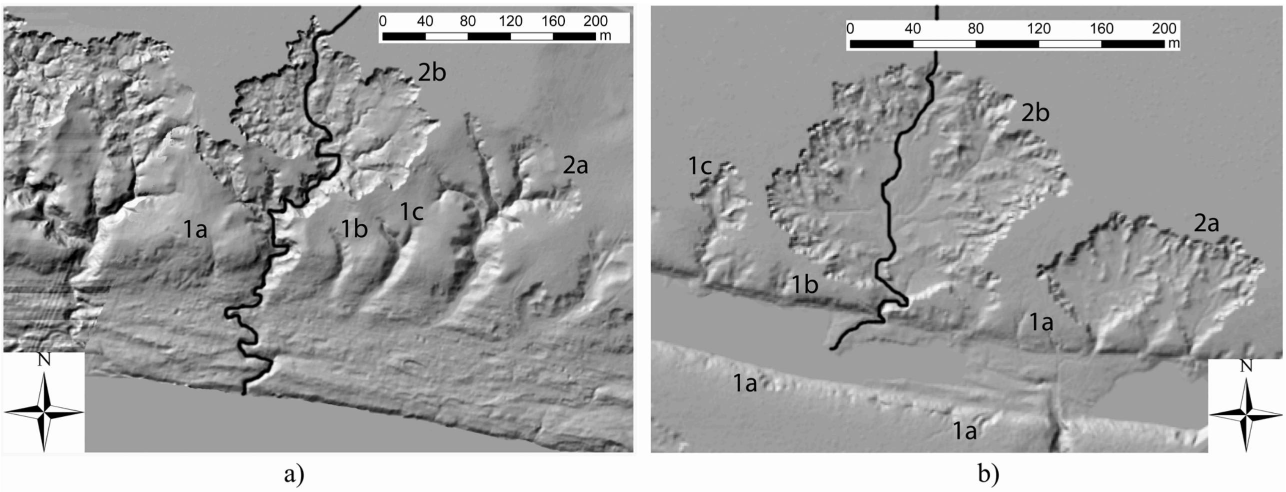

Figure 3. LiDAR DEM hillshades of alluvial gullies at different stages of evolution. Numbered gully labels in figures refer to stages of gully evolution: 1a to 1c are incipient gully stages, 2a and 2b are respectively, bounded and unbounded proximal gully stages.

Conclusions

Similar to analysis of Condon (1986) on the Victoria River in northern Australia, it is hypothesized that intense cattle grazing concentrated in the riparian zones of the Mitchell River during the dry season, in addition to fire regime modification and weed introduction, decreased perennial vegetation cover along hollows and the steep banks of river, both observed by Gilbert (1845) and Leichardt (1847). This land use change initiated a new larger phase of gullying and pushed the landscape across a threshold towards instability, which it was already close to as a result of the riverine landscape evolution over geomorphic time (Brooks *et al.* 2009). Once initiated on steep banks into dispersible sub-soils, alluvial gullies can rapidly progress in consuming and degrading the most productive part of the landscape, the riparian zone. The conceptual model of the evolution of these alluvial gullies prepared by Brooks *et al*. (2009) supports these hypotheses and describes their initiation, development, and potential stabilization over time. Overall, these data demonstrate the fragility of northern Australia's soils to land-use change and the potential for land-use change to cross erosional thresholds that permanently destabilize riparian landscapes.

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Mitigating global warming by improving terrestrial biotic carbon flux

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Abstract
There is global growing concern over increasing atmospheric CO_2, making Earth warmer and increasing frequency of extreme weather effects. Global carbons are pooled in five carbon pools and terrestrial carbon pool - a major pool - has a potential function to dampen increasing atmospheric concentration. The pedospheric mesofauna promote carbon sequestration by redistributing carbon through soil profile. A study of selected reclaimed overburden dumps, of different age, of mine spoils and a natural forest floor pedosphere ecosystem was carried out in Jharkhand. Quadrates of 1m x 1m x 0.05m were selected and soil corers were used for collection of samples for biotic and abiotic factors. Field data were analyzed using PAST software. We observed that there is a higher amount of soil organic carbon (SOC) in newly reclaimed over burden dumps (OBDs) than in older OBDs and natural forest, SOC and nitrogen are significantly correlated with soil mesofauna, there is significant positive correlation between above ground mesofauna and below ground mesofauna and there is higher soil mesofauna /kg in older reclaimed OBD and natural forest ecosystem. These results are indicative that there is a detrimental relationship between soil mesofauna and carbon sequestration – as there is lowering of SOC and soil organic matter/kg in older OBDs and forest.

Key words
Below-ground, arthropods, disequilibrium, non-equilibrium, ecosystem

Introduction
There is a growing concern that increasing levels of carbon dioxide in the atmosphere will change the climate, making the earth warmer and increasing the frequency of extreme weather effects (Schimel *et al.* 2001; Schlesinger and Licher 2001; Neff *et al.* 2002; Lal 2004; Bradford *et al.* 2007). The last two decades of the 20th century were the hottest in 400 years. It is expected that earth's mean temperature may increase by 1.5 – 5.8^{0}C in the 21st century (IPCC 2001). These climate changes are reportedly caused by emission of green house gases (GHGs) through anthropogenic activities (Lal 2007) and natural phenomena. Increase in GHGs coupled with increase in human population has cascading effect on the environment implicitly shifting ecosystem processes and functions (Running 2006; Westerling *et al.* 2006; Greene and Pershing 2007). Stakeholders, planners and environmentalist wish to stabilise the atmospheric abundance of carbon dioxide and other GHGs to mitigate the risk of global warming (Kerr 2007; Kintisch 2007b; Kluger 2007). According to Lal (2007) there are three ways of lowering carbon dioxide emissions; (1) Reduction in global energy use (2) Development of low or no carbon fuel (3) Sequestering CO_2 through natural or engineering techniques. The present paper's scope is limited to the third point and will emphasize the role of soil arthropods in carbon sequestration. The case study of rehabilitated mine spoils may be a model for the sink of atmospheric CO_2 for scaling up as there is no empirical data available to estimate the magnitude of carbon sequestration by soil insects. The environmental condition prevailing in the deep soil profile has detrimental role in the decomposition of soil organic matter (Gill *et al.* 1999; Gill and Broke 2002). The terrestrial ecosystems contain c2100 Gt of carbon (Schulze 2006), of which over two-thirds are stored in soils (Jobbagy and Jackson 2000; Amundson 2001). Part of this soil carbon pool is highly variable in space and time, while a large inert carbon pool may become active when exposed to new environmental condition (De Deyn *et al.* 2008). The large carbon storage capacity of soils suggests a potential function for soil to dampen increasing atmospheric concentrations (De Deyn *et al.* 2008). Soil carbon pools are balanced between carbon input via primary productivity and output via decomposition, leaching, burning and volatilization of organic compounds (Amundson 2001). The maximal potential of soil to sequestered carbon is determined by intrinsic abiotic soil factors, but soil carbon dynamics are essentially driven by biota and their interaction with climate (De Dedyn 2008). Soil mesofauna above ground as well as below ground promote carbon sequestration by redistributing carbon through the soil profile by channelling, mixing organic and mineral soil components, and by forming relatively stable soil aggregates and casts (De Deyn *et al.* 2008).

Materials and methods

The present study was carried out in the Kathara Coalfield area situated at 23^{O} 47' N latitude and 85^{O} 57' E longitude above 210 meters from mean sea level, in the district of Bokaro, Jharkhand, India. The average annual rainfall ranges between 157 cm - 195 cm and temperature oscillates between 2°C in winter to 45°C in summer. The soil is lateritic to acidic. Mine spoils are composed of coarse rocks generated due to deep coal mining operations and associated coal processing with low moisture and high temperature of the surface barring the natural growth of vegetation. Five sites were selected – three OBDs (spreading fly ash mixed with soil to give support for vegetation) of 5, 15, 30 years age of plantation, one derelict patch of fifty years age (approx.) with natural growth, and one natural forest. Samples of soil and litter arthropods – mesofauna were collected at regular monthly interval by standardized soil corers (0.05m) from randomized quadrates of 1m x1m covering an area of 100 sq. m. Each 250 - 300 gm litre sample was placed in a separate Berlese-Tullgren funnel and the arthropods were extracted. Data were analyzed using PAST software.

Results

We analyzed soil for its biotic and abiotic constituents. The abiotic physical properties of the soil are presented in Table 1. Results indicate that the SOC is higher in the newer reclaimed OBDs rather than in the older OBDs and in the forest (Figure 1). There are negative correlations between below-ground mesofauna and SOC (-0.5465), N (-0.5501), P (-0.956) and K (-0.9582), whereas there are positive correlations between above-ground mesofauna and SOC (0.5507), N (0.5447) P (0.0623) and K (0.0832).The linear regression between SOC and SOM expresses a highly (0.001) significant positive correlation (Figure 2). Principal component analysis (PCA) based on the soil mesofauna and nutrients (organic carbon, nitrogen, phosphate and potash) showed that at 63.05% (five years old, Figure 3) had strong association with soil mesofauna and SOC (component 1 and component 2). Similar trends were observed at 44.83%, 45.9%, and 39.28% for fifteen years, thirty years and fifty years respectively. SOM and nitrogen of the soil were also found positively correlated with each other in PCA.

Table 1. Soil profile of sites.

Sites	SOC/kg/h	N^{*}/kg/h	$P^{\#}$/kg/h	$K^{!}$/kg/h
I	1.5	808.3	41.7	165.8
II	1.4	766.7	40.2	160.5
III	0.9	503.3	38.9	161.1
IV	0.5	316.7	37.0	150.3
V	0.5	319.0	27.8	118.7

*N= nitrogen, #P= phosphate, !K= potash

Conclusions

Higher amount of SOC and SOM in freshly reclaimed OBDs of mine spoils may be because the ecosystem in disarray due to mining. The higher Shannon diversity index (DI) may be because ruderals and r-selects having higher reproductive capacity (Krebs 2004), which are exploratory meso-arthropods and are yet to adapt to a new habitat. On the contrary the lower DI and SOM in the older sites and forest reflect the presence of K - select species. The comparison of five to fifty years and a natural forest gives the impression that the system is in gradual change - switching from disequilibrium (Howard and Fisk 1911, Hembrom *et al.* 2008) to non-equilibrium (Krebs 2004; Sinha 2008) as no ecosystem is in equilibrium on the earth. Small disturbances put them in non-equilibrium state and it is the homeostatic and power of resilience that restores systems and provides opportunities for succession. But a larger disturbance exceeds the capacity of resilience of the system.

The results indicate that the systems have gone beyond their capacity of resilience, at the same time the available SOC and SOM are facilitating restoration processes. The positive correlation between above and below-ground arthropods (Hooper *et al.* 2000) is indicative of the restoration and improvement of habitat, which will improve the DI and vice-versa consequently. It was difficult to measure the missing carbon, in the pedosphere ecosystem as we were not able to account for all carbon in soil. But, the lowering of both soil mesofauna (below ground) and SOC are indicates that the residual carbon may be present in other supporting systems of the soil. Further, the microbes and mesofauna also immobilize organic carbon present in the soil as biomass (SOM). The SOM accumulates nutrients, which is important for above ground floral diversity (Devi and Yadav 2009). According to an estimate ten per cent of CO_2 flux to atmosphere through soil every year, which is more than ten times the CO_2, released from fossil-fuel combustion (Raich and Potter 1995). The large pool of mineralizable carbon in soil may be exploited by improved land management systems

(Kumar *et al.* 2009). Adopting conservation principles in general and improving soil meso arthropods taxonomic diversity in particular can mitigate GHG.

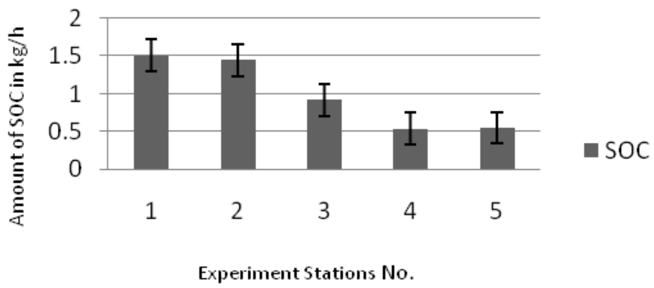

Figure 1. Soil organic carbon in different ages of reclaimed OBDs and natural forest of Jharkhand.

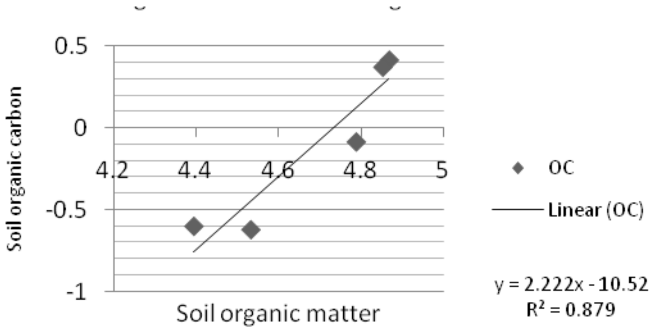

Figure 2. Linear positive correlation between soil organic matter and soil organic carbon.

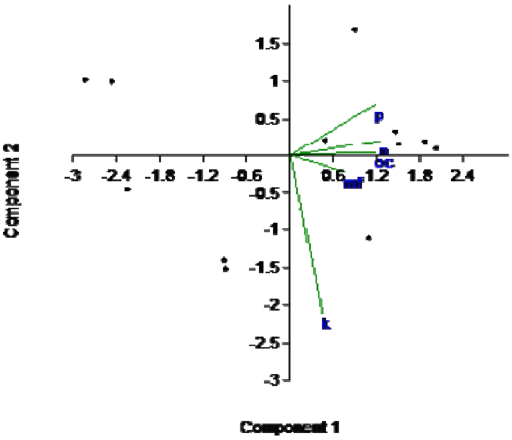

Figure 3. PCA between soil mesofauna OC, N, P, K of 5 years old OBD.

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Nutrient input through litter in riparian forest in different stages of ecological succession

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Abstract
This study was carried out from September 2008 to August 2009 and aimed to quantify the production, decomposition and the annual nutrient input from litter in a riparian zone in different stages of development and on different soils (Oxisol and Ultisol), within a representative plot of semideciduous mesophytic forest vegetation. Total litter produced in old and recent riparian forests on Oxisol was, respectively, 10.5 ton/ha/yr and 13.6 ton/ha/yr, while on Ultisol was 10.1 ton/ha/yr in old forest and 11.1 ton/ha/yr. The average time of renewal of forest litter in recent and in old forest was estimated at 0.77 yr and 0.57 yr. The amount of nutrients contributed by litter varied according to the total mass of litter produced, higher in recent forests. The concentration of macro and micronutrients contributed by litter showed the following order: Ca> N> K> Mg> P> S and Fe> Mn> Zn> B> Cu. The results indicated the important and significant role played by riparian forests in the cycling of nutrients and in restoration of soil fertility, conducting to the equilibrium and sustainability of the natural ecosystems.

Key Words
Soil fertility, soil chemistry, nutrient cycling, riparian forest, litter, rehabilitation of degraded areas.

Introduction
Riparian forests have important hydrological functions, such as protection of the riparian zone, sediment filtering, attenuation of xenobiotic molecules present in the flow from the surrounding agroecosystems, minimization of siltation, and control temperature of the aquatic ecosystem (Lima 1989, Martins 2001). The litter is particularly important for acting on the soil surface as a system of nutrient cycling, accumulating vegetal material that decompose and supply the soil and roots with nutrients and organic matter, which is essential in restoring soil fertility in degraded areas (Ewel 1976). The objectives of this experiment were to evaluate the rate of production and decomposition of litter from riparian vegetation in different stages of ecological succession, and estimate the annual nutrient input through litter.

Methods
Study areas
The study was conducted in two tracts of riparian ecosystems of 2 ha each, composed of two populations of semideciduous mesophytic forest, and an agroecosystem cultivated with sugar cane, which concentrically bordering the dam of Santa Lucia sugar mill, located in the São Paulo state, Brazil (22º18'00"S and 47º23'03" W, altitude 611 m). The forested areas differ in age [9 (recent forest - RF) and 18 years (old forest - OF)] and in soil type [Typic Hapludox (TH) and Arenic Hapludult (AH)], constituting four treatments (Figure 1).

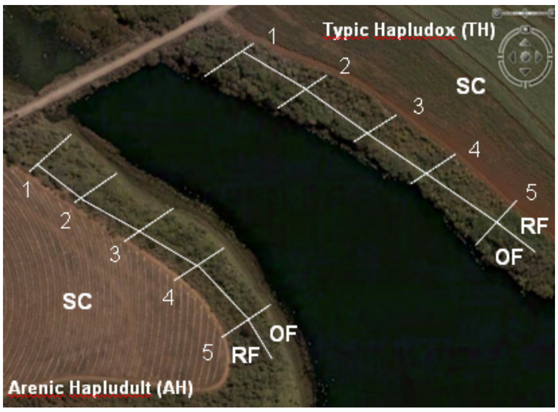

Figure 1. Representation of the transects in the experimental area (OF - old forest; RF - recent forest; SC - sugar cane) (Google Earth).

The climate is CWA (Köppen), i.e., mesothermal with hot and rainy summers and cold and dry winters. The average annual temperature is 21.4 º C and annual rainfall is 1448.8 mm (Brasil 1992).

Experimental characterization

For soil chemical and physical characterization, topsoil (0-0.2 m) and subsoil (0.2-0.4 m) samples were collected along five transects (Camargo *et al.* 1986; Raij *et al.* 2001) (Figure 1). The experimental area was divided into 20 plots of 100 m^2 (20 x 5 m) each (10 plots on the TH and 10 plots on the AH). In each plot were installed two circular collectors (0.5 m^2) for trapping litterfall at intervals of 30 days. The nutrient input from litter was estimated for macronutrients (N, P, K, Ca, Mg, S) and micronutrients (B, Cu, Fe, Mn, Zn) by the relationship between the biomass of total litter produced and the nutrient amount transferred by the biomass. The results were submitted to analysis of variance and differences between means determined by Tukey test at 5% level of probability. The rate of litter decomposition was estimated by the equation K = L/X, where K = decomposition coefficient, L = annual litter production, and X = average annual accumulated litter. The time required for decomposition of 50% of the litter (half life – $T_{0.5}$) was estimated by the equation $T_{0.5} = -\ln 0.5/K$. The average time needed to renew the stock of litter was obtained by the equation $tR = 1/K$ (Shanks and Olson 1961; Olson 1963).

Results

Calcium contents were highest in TH, decreasing from old to recent forest, but were low in both depth of AH. Phosphorus concentrations in the topsoil samples were probably due to the deposition, cycling and mineralization of organic matter, and were considered suitable for old and recent forests on the TH, but insufficient at the AH margin, where a less exuberant forest was observed. In general, micronutrients contents decreased from old to recent forest. With the exception of Zn in HA, no deficiency of micronutrients was verified (Table 1).

Table 1. Selected chemical attributes of topsoil and subsoil samples of Typic Hapludox (TH) and Arenic Hapludult (AH) collected under riparian forests.

Area	Depth (m)	P (mg/dm^3)	OM (g/dm^3)	pH ($CaCl_2$)	K	Ca	Mg	H+Al	Al	SB	CEC	V (%)	S	B	Cu	Fe	Mn	Zn
					(--------------$mmol_c/dm^3$						-------------)		(----------mg/dm^3					----------)
							Typic Hapludox (TH)											
OF	0-0.2	6.0	33.8	5.2	3.2	35.0	13.2	35.0	0.7	51.4	86.4	59.4	8.4	0.4	5.2	28.4	53.6	1.9
	0.2-0.4	3.6	25.6	5.0	2.3	26.0	10.4	37.0	1.0	38.7	75.7	50.8	5.8	0.3	4.8	23.6	53.8	1.1
RF	0-0.2	12.4	25.2	5.0	2.2	25.0	8.2	42.2	1.5	35.4	77.6	46.3	8.2	0.2	4.4	17.0	49.4	1.1
	0.2-0.4	6.0	20.8	4.9	0.9	20.8	5.0	41.4	1.6	26.7	68.1	39.5	11.8	0.1	3.6	13.6	38.2	0.6
							Arenic Hapludult (AH)											
OF	0-0.2	4.3	24.9	4.7	3.8	19.3	9.4	33.9	2.3	32.5	66.4	48.5	9.7	0.7	2.1	63.8	42.3	1.7
	0.2-0.4	1.3	11.1	4.6	1.9	16.3	6.2	33.3	2.7	24.4	57.7	40.7	9.0	0.3	1.5	30.9	34.2	0.5
RF	0-0.2	5.0	16.3	4.6	3.2	16.6	5.4	35.1	5.6	25.2	60.3	40.9	15.1	0.4	2.1	42.8	20.9	0.5
	0.2-0.4	2.5	11.2	4.4	2.0	15.2	5.1	44.4	9.6	22.3	66.7	34.7	18.7	0.5	1.6	30.5	27.1	0.4

OF – old forest; RF – recent forest; OM – organic matter; SB – sum of bases; CEC – cation exchange capacity; V - level of base saturation

Clay contents varied from 140 g/kg (AH) to 565 g/kg (TH). The amount of soil organic matter (SOM) decreased in depth in all treatments and varied from 11.1 g dm^{-3} to 33.8 g dm^{-3} (Table 1). For 30 non-cultivated topsoil samples collected in areas of native vegetation, Soares and Alleoni (2008) obtained a wide range of SOM from 6.6 g/kg to 213.4 g/kg, while Bayer and Mielniczuk (1997) verified that SOM content of an Ultisol was reduced from 31 g/kg to 18 g/kg due to successive cultivations. Both reduction in the cultivation intensity and forest development are contributing for organic matter addition through litter. The old forest showed higher levels of organic matter than recent forest, pointing to greater equilibrium of old forest in relation to the stability of organic matter and nutrient cycling. Concomitantly, the cation exchange capacity (CEC) under continuous cultivation usually decreases with time due to the reduction of topsoil organic matter (Sanchez *et al.* 1983; Cerri *et al.* 1991). This was evident in the topsoil and subsoil samples from the TH and AH, which showed decrease of CEC values from the old to the recent forest in both layers (0-0.2 m and 0.2-0.4 m). Both SOM and CEC are useful indicators for soil quality and sustainable land management. The mean contribution (per gram) of the SOM for the soil CEC is 1.64 $mmol_c$, i.e., 44 times

higher than the contribution of the clay fraction (Soares and Alleoni 2008). The increase of SOM was an important indicative of soil fertility recovery and land restoration. The total litter produced in old and recent riparian forests on TH was, respectively, 10.5 ton/ha/yr and 13.6 ton/ha/yr, with significant statistical difference. No significant statistical difference was observed for total litter produced by different forests on AH (10.1 ton/ha/yr in old forest and 11.1 ton/ha/yr in recent forest). Tropical forests have continued production of litter throughout the year, and the quantity produced at different times depends on the type of vegetation considered (Leitão-Filho 1993). More disturbed areas have very large number of pioneer species, with fast growing, short cycle and higher nutritional requirements, investing heavily in biomass production in a short time. On the other hand, the less modified areas have a higher number of climax species, with lower production of biomass and nutrients requirement (Martins and Rodrigues 1999). Thus, differences in litterfall between near sites may be related to different degrees of disturbance that are found within the same forest type (Werneck *et al.* 2001). It is believed that the metabolic activity of plants in recent forests is higher than in the old forest, due to lower floristic age which would allow greater biomass production.
Generally, significant statistical difference was verified for N, S and Ca inputs through litter in old and recent riparian forest with respect to soil type, due to the higher vegetal biomass of the forests established on the TH (Table 2).

Table 2. Macronutrients annual inputs (kg/ha) through litter of riparian forests.

Soil	N		P		K		Ca		Mg		S	
	OF	RF	OF	RF	OF	RF	OF	RF	OF	RF	OF	RF
TH	188.7 Aa	278.0 Ba	13.4 Aa	15.5 Aa	27.8 Ba	43.9 Aa	215.8 Ba	265.2 Aa	20.7 Ba	31.4 Aa	13.9 Ba	18.5 Aa
AH	159.6 Bb	183.9 Ab	11.4 Aa	12.9 Aa	34.2 Aa	39.5 Aa	164.6 Bb	219.8 Ab	24.7 Ba	30.5 Aa	11.0 Aa	13.5 Ab

TH – Typic Hapludox; AH – Arenic Hapludult; OF – old forest; RF – recent forest
Different capital letters in the line for the same nutrient indicate statistical differences according to Tukey test (p<0.05)
Different lowercase letters in the column for the same riparian forest indicate statistical differences according to Tukey test (p<0.05)

The concentration of nutrients contributed by litter followed the order: Ca> N> K> Mg> P> S. Morellato (1992) observed an annual addition through litter of 206 kg N/ha, 11.2 kg P/ha, 37.8 kg K/ha, 269.2 kg Ca/ha and 29.9 kg Mg/ha. In our experiment, we obtained the following range of N, P, K, Ca and Mg, consecutively: 159-278; 11.4-15.5; 27.8-43.9; 164.6-265.2 and 20.7-30.5 kg/ha /yr (Table 2).
For micronutrients, significant statistical difference was verified for Fe and Mn inputs through litter in old and recent riparian forest with respect to soil type (Table 3).

Table 3. Micronutrients annual inputs (kg/ha) through litter of riparian forests.

Soil	B		Cu		Fe		Mn		Zn	
	OF	RF	OF	RF	OF	RF	OF	RF	OF	RF
TH	0.35 Ba	0.47 Aa	0.28 Aa	0.28 Aa	106.1 Aa	84.2 Ba	1.76 Ba	2.32 Aa	0.50 Aa	0.54 Aa
AH	0.33 Aa	0.34 Aa	0.21 Aa	0.21 Aa	53.8 Bb	77.2 Aa	1.44 Aa	1.43 Ab	0.42 Aa	0.46 Aa

TH – Typic Hapludox; AH – Arenic Hapludult; OF – old forest; RF – recent forest
Different capital letters in the line for the same nutrient indicate statistical differences according to Tukey test (p<0.05)
Different lowercase letters in the column for the same riparian forest indicate statistical differences according to Tukey test (p<0.05)

The concentration of nutrients contributed by litter followed the order: Fe> Mn> Zn> B> Cu.Vital *et al.* (2004) observed annual addition through litter of 18.3 kg Fe/ha, 5.6 kg Mn/ha, 0.33 kg B/ha, 0.22 kg Zn/ha and 0.15 kg Cu/ha. Compared with the study of Vital *et al.* (2004), our results indicated that the inputs of B, Fe, Zn and Cu were higher; whereas Mn was lower (Table 3). It is evident that, in the recent forest, the input of macronutrients and micronutrients through litter was higher than in the old forest; regardless of soil type (Tables 2 and 3). The decomposition coefficient (K) of old forest was lower than of recent forest. In semideciduous mesophytic forest, this parameter ranges from 1.2 to 1.9 (Morellato 1992). The renewal average time of forest litter in recent forest on TH was estimated at 0.53 years, i.e., 193 days. In the recent forests on TH and on AH, the time required for decomposition of 50% of the litter was estimated at 0.37 years and 0.43 years, i.e., 135 days and 157 days, respectively. These values are close to 150 days reported by Vital *et al.* (2004) (Table 4).

The results indicate rapid release and recycling of nutrients by the vegetation. Decomposition rate differences of litter from tropical forests could be attributed to the type of vegetation, quality of material, the soil microbiota activity, and environmental conditions, especially temperature and humidity.

Table 4. Decomposition rate (K), time required for decomposition of 50% of the litter ($T_{0.5}$) and mean litter renewal (tR) for riparian forests.

Soil	K		$T_{0,5}$ (years)		tR (years)	
	OF	RF	OF	RF	OF	RF
TH	1.29	1.89	0.54	0.37	0.78	0.53
AH	1.31	1.63	0.53	0.43	0.76	0.61

TH – Typic Hapludox; AH – Arenic Hapludult; OF – old forest; RF – recent forest

Conclusions

The amount of nutrients contributed by litter varied according to the total mass of litter produced, in higher quantity in the recent forests in comparison with the old forest, regardless soil type. In the recent forests, the cycling of nutrients occurred more quickly. Since in the old forests, this time was larger, providing nutrients more slowly, indicating a better equilibrium and sustainability of the forest. The results indicated the important and significant role played by riparian forests in the nutrients cycling and in soil fertility restoration.

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Nutrient status of cocoa *(Theobromae cacao)* in Papua New Guinea: results from a survey

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Abstract
Cocoa is the primary cash crop in most coastal areas of Papua New Guinea (PNG), supporting an estimated 150,000 households. Smallholders produce most of the crop, but their yields are low, at about 10% of the maximum. Low yields have been attributed to many factors, but the possibility of nutrition-related limitations to productivity has not been examined in detail. Here we report on a survey of 63 cocoa blocks across the country. Based on leaf analyses, N and Fe deficiencies appear to be very widespread, with 95 % of sampled blocks falling below the critical level for N and 89 % for Fe. P deficiencies were encountered in ~25 % of the blocks sampled. Leaf Mg concentrations were adequate in most blocks in most provinces, except East New Britain, where 64 % of the blocks sampled were deficient. Deficiencies of K, Ca, Mn, B, Cu and Zn were encountered in 2 -15 % of sampled blocks. There were significant relationships between leaf and soil contents of K, Ca, Mg and P. There is a clear need to further examine nutrition-related limitations to productivity in PNG, including establishment of more reliable critical levels for leaf nutrient concentrations.

Key Words
Critical level, deficiency, leaf nutrient content, nutrient cycling, soil fertility, tree crop.

Introduction
Cocoa is a major cash crop in Papua New Guinea (PNG), contributing substantially to the national economy in terms of employment and foreign exchange earnings (ranks third after oil palm and coffee). PNG currently exports an average of 43,000 t of cocoa annually from an estimated 100,000-130,000 ha, bringing in export earnings of about US$ 62.1 million annually. Smallholders (approximately 150,000 households) produce over 80% of PNG's cocoa, with the balance coming from plantations. Smallholder yields are very low, generally in the range 0.3 - 0.4 t/ha of dry bean annually. Yield potential is much higher, with annual yields of up to 4.4 t/ha having been observed in research trials, and up to 2.5 t/ha being obtained in plantations. Low yields have been attributed to many factors, including labour shortages, low levels of block maintenance (eg. pruning, shade control and weeding), lack of appropriate agronomic knowledge, land shortages and cocoa prices. Most cocoa blocks in PNG can be classified as 'senile' (>8 years old), having few accessible ripe pods and low management inputs (Curry *et al.* 2007). The possibility of nutrition-related limitations to productivity has been raised in the past but not examined in detail. Most of the production comes from land that has been cropped for 15 years or more with little or no fertiliser inputs. The objective of this study was to assess the likely nutrition-related limitations to production across the main growing areas of the country.

Methods
Between April and November 2007, 63 sites across the country were surveyed (Figure 1). The survey consisted of interviews with the growers, sampling of soil and leaf tissue, and in some cases, pods and husks. Of the 63 sites surveyed, 48 were on smallholder blocks, 6 were on plantations, 8 were in Cocoa Coconut Institute (CCI) trials and 1 was on a potential cocoa site. By province, 11 were in East New Britain, 9 in Autonomous Region of Bougainville, 9 in New Ireland, 8 in Madang, 8 in East Sepik, 6 in Morobe, 6 in Northern, 4 in West Sepik and 2 in the Jimi Valley of Western Highlands. At each site, a plot of 42 (6 x 7) cocoa trees was selected for sampling. The plot was assessed for tree health and general maintenance and the grower was interviewed about block history and maintenance. An attempt was made to calculate yields, but it was not possible to make reliable estimates for most blocks as most smallholders do not keep production records. Leaves were sampled from 20 trees distributed evenly throughout the 42-tree plot. The leaves chosen (2 per tree) were the third leaf of a recently hardened leaf flush at mid-canopy height. Leaves were

dried, weighed and ground, and a composite sample was prepared for each site. Cocoa pods were sampled from 8 sites. At each site, 10 ripe pods were picked, with no more than one pod being picked per tree within the 42-tree plot. The beans and husks were separated, weighed, dried and weighed again. They were then ground and mixed, and composite sub samples prepared. Soil samples were taken at depths of 0-0.15, 0.15-0.30, 0.3-0.6 and 0.6-0.9 m depth, using an auger. Samples were taken 1m from the tree trunk at trees distributed evenly throughout the 42-tree plot. The shallowest depth increments were sampled at 9 trees and the deeper increments at 5 of those trees, and one composite sample was prepared for each depth increment at each site. Plant tissue samples were analysed for macro and micro elemental contents. Assessments of deficiency or otherwise were made using the leaf concentration values noted by Fahmy (1977). These values have only ever been described as 'tentative' because they have not been verified by trials in PNG. Furthermore, it should be pointed out that foliar analysis has been less useful for diagnosis and management of nutrition problems in cocoa than for other crops. That is because leaf age and light intensity usually over-ride the nutritional effects on leaf nutrient composition except when there are marked deficiencies (Wessel 1985). Soil samples were analysed for field texture and a range of parameters related to chemical fertility. A subset of subsoil samples were analysed for mineralogy by x-ray diffraction and x-ray fluorescence.

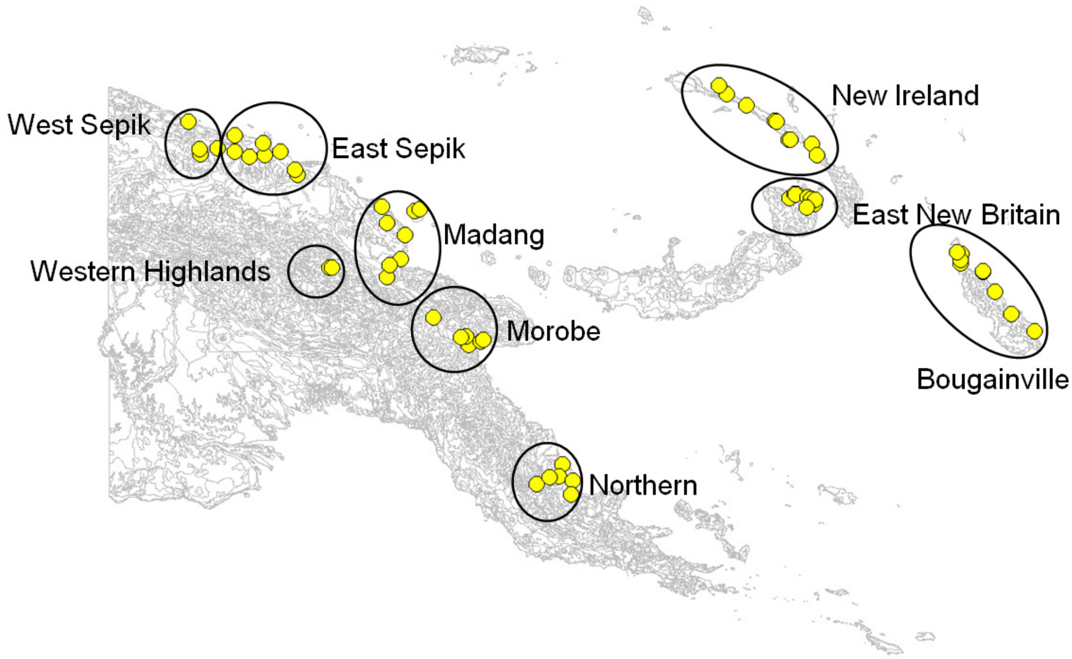

Figure 1. Map of Papua New Guinea showing the sampling locations and province names.

Results and Discussion

Based on leaf analyses, some nutrient deficiencies were widespread (Figure 2). For N, 95 % of sampled blocks fell below the critical level. Leaf N:P ratios were low, with a mean of 10.4, indicating a deficiency of N relative to P at most sites. At only 10% of sites did leaves appear deficient in K. This was surprising as K deficiencies have been reported, particularly on coralline soils, and many sites in this study had low soil exchangeable K contents and high ratios of exchangeable Ca:K or Mg:K. Two possible reasons for the discrepancy are a) the critical leaf value is not realistic, or b) K deficiency is not expressed because another deficiency (eg. N) is limiting. At most sites (89%), leaves had deficient or suboptimal Fe concentrations compared to the published tentative critical levels. P deficiencies were encountered in ~ 25% of the blocks sampled and were distributed across all provinces. Leaf Mg concentrations were adequate in most blocks in most provinces, except East New Britain, where 64 % of the blocks sampled were deficient. Deficiencies of K, Ca, Mn, B, Cu and Zn were encountered in 2 -15 % of sampled blocks. From leaf measurements, it appeared that leaves expand to near full size early in development and then accumulate dry matter as they mature. Nutrient exports, calculated from pod analyses, were generally within the range measured elsewhere (Hartemink 2005), but N exports were relatively low (18-22 kg/t dry beans) and K exports (11-15 kg/t dry beans) were higher than previously reported values.

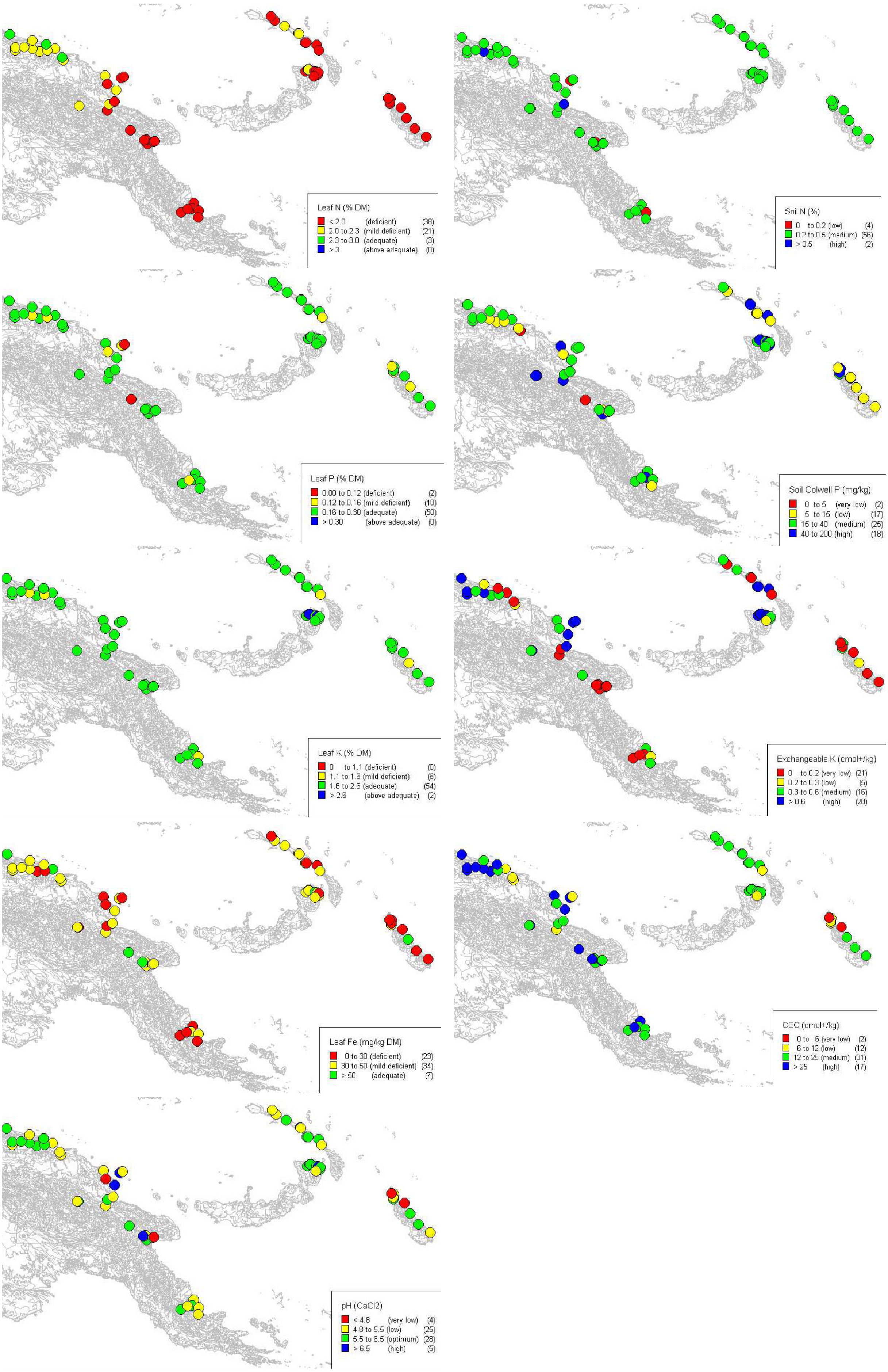

Figure 2. Map of Papua New Guinea showing leaf nutrient contents and soil properties (0 – 0.15 m depth).

Virtually all soil profiles sampled in this survey progressed from loam topsoil to sandy loam or clayey loam subsoils. Soil mineralogy was diverse, from little weathered profiles dominated by glass or feldspar to highly weathered profiles dominated by halloysite. Allophane content was <11% over all sites and was significantly correlated with phosphate buffer index. Most sites had reasonably high soil CEC, pH and organic C contents. Sites with high soil exchangeable K content had high leaf K contents and sites with high CEC and exchangeable Ca had high leaf Ca and B contents. All sites with low concentrations of K or P in the leaves had low soil exchangeable K or extractible P contents, respectively. There was also a significant correlation between leaf Mg concentration and the ratio of soil exchangeable Mg:K.

There was a wide range of crop species cultivated in the surveyed cocoa blocks, although *Gliricidia* was the most common shade tree. However, no relationships were evident between cocoa nutrition status and management factors, including shade and the presence of legumes as shade trees or cover crop.

Conclusions

Nutrient deficiencies in cocoa, especially N, are common in PNG and are partly related to soil properties and geographical location. If the widespread N deficiencies were to be overcome it could be expected that the extent and severity of other nutrient deficiencies would increase substantially. The current assessment is based on tentative critical levels for leaf nutrient concentrations and there is a clear need to produce reliable critical levels based on manipulative trials.

Acknowledgements

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Potential of Quesungual agroforestry system as a land use management strategy to generate multiple ecosystem services from sub-humid tropical hillsides

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Abstract
According to the Millennium Ecosystem Assessment, ecosystem services (ES) are the benefits people obtain from ecosystems, including provisioning services such as food and water; regulating services such as flood and disease control; cultural services such as spiritual, recreational, and cultural benefits; and supporting services such as nutrient cycling that maintain the conditions for life on Earth. The Quesungual Slash and Mulch Agroforestry System (QSMAS) has been suggested as a land use management strategy to generate multiple ES on hillsides of sub-humid tropics. Research studies conducted in Honduras from 2005 to 2007 showed that the production practices applied in QSMAS generates ES through beneficial effects on the soil plant-atmosphere continuum. Specifically, QSMAS contributes to food security through sustainable maize and common bean production under sub-humid conditions on steep slopes, by improving crop water productivity and soil quality, compared to the traditional slash and burn (SB) system. Additionally QSMAS is eco-efficient through the use of renewable natural resources, and also provides ES by reducing deforestation, soil erosion and global warming potential compared to the traditional SB system.

Key Words
Smallholder agriculture, sustainability, resilience, payment for environmental services.

Introduction

Within the terrestrial ecosystems, the soil is the main provider of environmental services (ES). The soil is a living system essential to sustain biological productivity, air and water quality, and plant, animal and human health (MEA 2005). Unfortunately, soil degradation is a severe problem for food production in rural areas, particularly in developing countries. Therefore, it is necessary to contemplate strategies for land management representing the best possible communion between generation of multiple services while preserving the natural capital (Lavelle 2008). The Quesungual Slash and Mulch Agroforestry System (QSMAS) is a smallholder production system that makes use of a group of technologies for the sustainable management of vegetation, water, soil and nutrient resources in drought-prone areas of hillside agroecosystems of the sub-humid tropics. The system was developed in southwest Honduras, Central America, by improving native farming practices with the participation of local farmers and technicians of Food and Agriculture Organization (FAO) and other national and international institutions. The system is based on planting annual crops (maize, common bean, and sorghum) with naturally regenerated trees and shrubs. QSMAS is being practiced by smallholders in Honduras, where the system has been successfully adopted by over 6,000 resource-poor farmers on 7,000 ha. This resulted in a locally recognized suitable alternative to the traditional slash and burn (SB) system, with biophysical and socioeconomic benefits at multiple scales ranging from farm level (increased crop water productivity, food security) to landscape (increased amount and quality of available water). The set of technologies responsible for the success of QSMAS can be summarized in four basic principles of conservation agriculture that contribute synergistically to its superior performance: (1) no SB, but through the management of natural vegetation; (2) permanent soil cover, through the continual deposition of biomass from trees, shrubs, and weeds, and through crop residues; (3) minimal disturbance of soil, through the use of no tillage, direct seeding, and reduced soil disturbance during agronomic practices; and (4) efficient use of fertilizer, through the appropriate application of fertilizers. The main objective of this research work was to determine the key principles behind the biophysical resilience of QSMAS and its capacity to sustain crop production and alleviate water deficits on steeper slopes with greater risk for soil erosion.

Methods

The performance of QSMAS was studied in southwest Honduras, within the Lempa River upper watershed department (district) of Lempira, from 2005 to 2007. Mean annual (bimodal) precipitation is ~1400 mm falling from early May to late October, with a long dry season of up to 6 months. Field plots were established to compare 5 main treatments (replicated on three different farms): QSMAS of three different ages (<2, 5-7 and >10 years-old), the traditional SB system, and secondary forest (SF) as a reference land use system (LUS). Annual management of QSMAS plots included slashing and mulching through pruning of trees and through crop residues while SB plots were managed through slashing and burning of native vegetation, before the onset of the rainy season. Maize and common bean were established in the early (late May) and later (late August) part of the rainy season, respectively, and managed following the timing, spatial arrangement and management practices that are commonly used in the region for the production systems under comparison. The four production system treatments (QSMAS of different ages and SB) were split in order to apply a fertilizer treatment (addition vs. no addition). In the fertilized treatments, the maize received 101 kg/ha of N and 55 kg/ha of P, while the common bean received 46 kg/ha of N and 51 kg/ha of P. Studies included monitoring and analysis of soil water dynamics, crop water productivity (CWP), greenhouse gas (GHG) fluxes, carbon sequestration and global warming potential (GWP).

Water infiltration and runoff were measured through rainfall simulation for 30 minutes using two intensities (80 and 115 mm/h). Soil water content was determined through soil sampling at three depths (0-10, 10-20 and 20-40 cm). Susceptibility of the soil to erosion was assessed in erosion plots (5 m length x 1.5 m width) over 3 years. Soil losses were determined through the comparison of the indices of soil erodibility K-USLE and Ki-WEPP corresponding to the Universal Soil Loss Equation (Wischmeier and Smith 1978) and to the Water Erosion Prediction Project (Nearing *et al*. 1989), respectively. Nutrient losses through erosion were quantified by determining total contents of N, P, K, Ca and Mg from samples of eroded soils. Water quality was assessed through the determination of NO_3^-, NH_4^+, total P, and PO_4^{-3} in samples collected at 45 DAP. Both eroded soil and water samples were collected in erosion plots in 2007. CWP, expressed as kg of grain produced per m^3 of water used as evapotranspiration, was calculated using the crop yield and soil water data obtained in 2007 and by estimating the evapotranspiration (ET) according to the method of Penman and Monteith (FAO 1998).

Annual GHG fluxes between soil and atmosphere were monitored using the closed chamber technique as described by Rondón (2000). At the beginning of the study, 4 PVC rings (height 8 cm, ø= 25 cm) were located in the experimental plots. In every chamber and at each sampling date (16 dates), 4 air samples were taken at 0, 10, 20 and 30 minutes, after installing the chamber (height 10 cm, over the PVC ring). Air samples were extracted from the closed chamber using a syringe with an adapted valve and then introduced into glass containers (pre-vacuumed vials by freeze drying). N_2O and CH_4 concentrations were determined in the laboratory, using a Shimadzu GC-14A gas chromatograph, equipped with FID (flame ionization detector) and ECD (electron capture detector) for methane (CH_4) and nitrous oxide (N_2O) detection, respectively. For CO_2 concentration, we used a Qubit Systems S151 gas analyzer, with infrared technology. GWP of the different LUS was calculated by using CH_4 and N_2O fluxes between soil and atmosphere, and C stocks from soil and tree biomass. For the traditional SB system direct emissions of CO_2, CH_4 and N_2O, from the biomass burning were also included. GHG fluxes of each LUS were multiplied by the global warming potential value, corresponding to the GHG (CO_2=1, CH_4=72 and N_2O=289) in a 20 year time horizon (IPCC 2001).

An emergy (from "embodied energy", a measure of the total energy used to make a product or service) evaluation was conducted as in Diemont *et al* (2006) to quantify resource use and system sustainability, using data from plots and relationships (energy input per unit of energy output) reported in other studies. The Environmental Loading Ratio (ELR, a measure of ecosystem stress due to a production activity) was given by the ratio from purchased and nonrenewable local inputs, to the emergy from renewable resources.

Results

Evaluation of water dynamics at the middle of the rainy and dry seasons of 2007 showed a lower infiltration and higher runoff in SB system. During the rainy season, SB had the lowest infiltration (29.8 mm) and highest runoff (12 mm); in contrast, QSMAS >10 years had the highest infiltration (38.5 mm) and lowest runoff (4.8 mm). During the dry season differences between treatments in infiltration and runoff were small. Infiltration for 30 minutes ranged from around 44 mm in both QSMAS treatments to 41.9 mm in SB. Runoff ranged from 0.91 mm in QSMAS to 2.4 mm in SB. In 2007, precipitation and ET were 1005 and 491 mm in the early part of rainy season, and 419 and 272 mm in the later part, respectively. In the early part of the rainy season available soil water (0-40 cm soil depth) varied between 0.09 and 0.104 m^3/m^3, with QSMAS <2 and QSMAS 5-7 and was 10% and 16% higher, respectively, than in SF. In the later part of the rainy season the amount of available soil water varied between 0.11 and 0.127 m^3/m^3 in SB and QSMAS <2, respectively. The mean value of available soil water content (0-40 cm) in QSMAS systems (average of the three different ages) was significantly greater than that of the SB system, suggesting increased availability of water for crop growth. These improvements in QSMAS were related to changes in soil porosity due to increases in mesoporosity (30%) and macroporosity (19%), and decreased the soil bulk density. This increased the plant available soil water storage capacity and availability of water for crops in the dry season, and increased the capture of rainfall at the beginning of the rainy season. The highest soil loss occurred in 2005, and was markedly higher in SB followed by QSMAS and SF. The same trend was observed in 2006 and 2007, although differences were greater in 2005 due to higher rainfall intensity and to the recent conversion of SB plots from SF that resulted in bare soil and therefore higher susceptibility to erosion. Total soil losses over the 3 years from SB were 5.6 times greater than from the three QSMAS treatments, and 22 times greater than from SF. As a result, the SB system had the highest nutrient losses (kg/ha) of N (9.9), P (1.3), K (6.9), Ca (22.8) and Mg (24.2), while SF had the lowest losses of N (1.7), P (0.2), K (1.2), Ca (2.6) and Mg (2.7). Water quality was poorest in the SB system, with highest concentration (mg/L) of total P and PO_4^{-3} (2.30 and 0.29, respectively), and was much better in QSMAS >10 (0.18 and 0.25, respectively). SB also had the highest concentration of (mg/L) of NO_3^- and NH_4^+ (7.97 and 0.70, respectively), while QSMAS 5-7 had the lowest concentration of NO_3^- (6.13) and QSMAS >10 of NH_4^+ (0.24). SF had values of 0.65 for P, 0.43 for PO_4^{-3}, 4.73 for NO_3^-, and 0.92 for NH_4^+.

There was no interaction between LUS and fertilizer treatment on CWP. CWP (kg grain/m^3) for maize was greatest in fertilized systems of QSMAS <2 (0.48) and least with QSMAS >10 (0.18). In plots with no fertilizer application, the highest CWP was observed with QSMAS <2 (0.26) and the lowest with SB (0.10). In both fertilized and non-fertilized systems, CWP for common bean was greatest in QSMAS <2 (0.32 and 0.27 kg grain/m^3, respectively) and least with SB (0.10 and 0.07 kg grain/m^3, respectively). Fertilization increased CWP of maize (by 92%) and common bean (by 23%). These results may reflect adequate available soil water during the maize crop (from sowing to physiological maturity) in the early part of the rainy season, as precipitation was higher than ET. In the case of common bean grown in the later (drier) part of the rainy season, available water content in the soil decreased from flowering to physiological maturity, with lower precipitation than ET and therefore with a negative water balance. Under these conditions, QSMAS showed greater available water content in soil that resulted in greater grain yield and CWP.

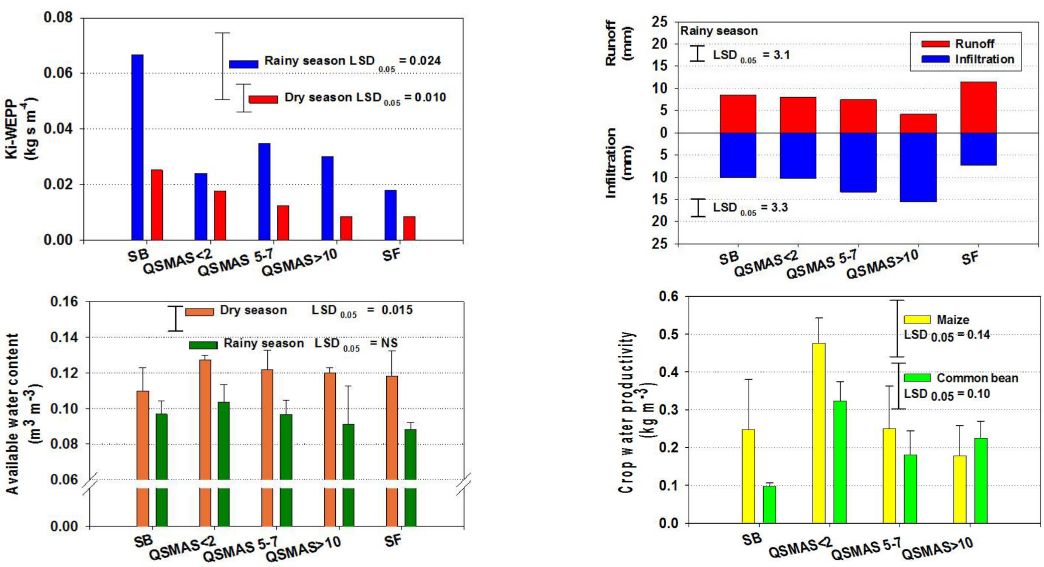

Figure 1. Provisioning services provided by QSMAS: improved water cycling through reduced susceptibility to erosion (top left), increased infiltration and decreased runoff (top right) and improved soil water storage capacity (bottom left), and improved food security through enhanced crop water productivity (bottom right).

C stocks were higher in SF and QSMAS, with higher accumulation in SF for aboveground C (C in trees and shrubs) and in QSMAS >10 for belowground (soil organic) C (Figure 2). The SB system could generate higher annual losses of above ground C due to burning, while young QSMAS plots (<2 and 5-7 years old) generate some losses of below ground C. QSMAS also had a much lower GWP (10.5 Mg Equiv. CO_2) than SB traditional system (40.9 Mg Equiv. CO_2). SF had a very low GWP (1.14 Mg Equiv. CO_2) (Figure 2). Based on the current adoption of QSMAS and consequent regeneration of SF in the Lempira department where QSMAS is practised and projecting its impact on GWP for a period of 20 years, it is estimated that the adoption of QSMAS will result in a decrease of 0.10 Tg Equiv. CO_2 compared to SB. Higher C stocks in soil and the aboveground tree biomass indicate a gradual accumulation of C in SF and QSMAS >10. According to the emergy evaluation SF and QSMAS had less environmental impact than SB (highly affected by levels of soil erosion) as noted in the ELR with values of 0.63, 0.14, and 0.02, respectively.

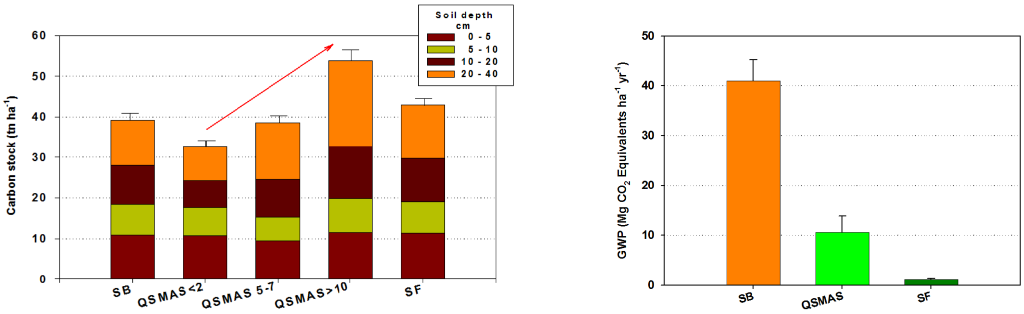

Figure 2. Regulating services provided by QSMAS: reduced global warming potential through improved C accumulation (left) and lower methane emission (right).

Conclusions

The results indicate that the production practices used for managing QSMAS have beneficial effects on the soil-plant-atmosphere continuum, soil quality, landscape and the environment. Compared to SB system QSMAS is eco-efficient through the use of renewable natural resources, and also provides ecosystem services including: (1) Provisioning services: food security through improved crop water productivity and yields at lower costs; and improved water cycling through reduced runoff, erosion, water turbidity and surface evaporation, and increased infiltration, soil water storage capacity and use of green water; (2) Regulating services: reduced global warming potential through lower methane emission and improved C accumulation; (3) Supporting services: mitigation of soil degradation through improved structure, biological activity, organic matter, nutrient cycling and fertilizer use efficiency, and restoration and conservation of biodiversity; and (4) Cultural services: improved quality of life through the regeneration of the landscape. Potential on the payment for environmental services provided by QSMAS could enhance its attractiveness to local and national authorities in countries with policies to protect ecosystems in the face of climate change.

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Resalinization and low productivity of recently reclaimed salt – affected soils

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Abstract
The objective of this investigation is to find out the reason/s of resalinization of recently reclaimed salt-affected soils. Columns of saline-sodic soils were subjected to leaching by maintaining 10 cm deep layer of water on the soil surface. The leaching process was terminated when electrical conductivity (EC) of the effluent dropped just below 4 dS/m. It was found that 40 % of the total sodium initially present in these soils, remained after the termination of leaching process. This high level of sodium in soil was considered the main reason for resalinization of reclaimed soils in Iraq at which EC of the drain water was used to determine extent of reclamation process. Resalinzation process was attributed to the process of equilibration between solid and solution phases of soil. Accordingly total sodium in soil rather than EC of the effluent (EC of the drain water) is suggested as the major parameter to be considered for determining the reclamation boundary, rather than EC of the drainage water of the field under consideration.

Key Words
Leaching, effluent, total sodium, electrical conductivity, nutrient loss, sodic soil.

Introduction
Reclamation of salt –affected soils consists of salt removal process and lowering salty water table (Balba 1972). It is well known that most reclaimed soils in Iraq are resalinized again in relatively short period of time after cropping. Resalinization process was thought as a consequence of defect in water and salt balance or due to negligence in maintaining active drainage system (Alzubaidi 1989). Others had attributed the problem of resalinization to bad management of farmers (Reports of State Organization of Soil and Reclamation). The majority of salt-affected soils in Iraq are saline – sodic soils which contain high amounts of sodium (Na) along soil profile (Muhawish 1995). Previously, investigators used disturbed soil columns to evaluate leaching process in term of salt movement and distribution along soil profile. Electrical conductivity or chloride ion was used as the index for salt movement and behavior during leaching. Accordingly, quantity of sodium salts remained in soil after the termination of leaching process is completely overlooked in previous studies. Therefore, in this study total sodium content will be used as an indicator for determining the extent and completion of reclamation process. Na displacement in soils differed in salt content, chemical composition and sodium adsorption ratio will be studied as well. These data will be used to find out the reason/s of resalinization of recently reclaimed salt-affected soils in Iraq which is the main objective of this study.

Materials and methods
Two most common salt-affected soils locally named as Shura and Sabakh were used in this study. Shura soil is characterizes by white salt crust on the surface. Sabakh soil on the other hand, is characterizes by accumulation of deliquescent salts on the surface. Soils are classified as Typic Torrifluvent. Samples of both soils were collected from Salman project 20km south of Baghdad. Samples were collected along soil profile starting from the surface down to 80 cm, including the following layers 0-10, 10-20, 20-30, 30-60, and 60-80 cm. Some of soil properties are shown in Table 1. The soil samples were transferred in the same sequence as in field into galvanized iron columns of 100 cm length and 30cm diameter. Soil columns were subjected to conventional leaching under constant water head (10 cm depth) using water collected from Tigris river (EC=0.57 dS/m, pH =7.0).

Leaching process continued until the EC of percolated water dropped below 4 dS/m. Effluent of each column was collected throughout leaching periods to measure pH, EC, and soluble ions. Soil samples along soil columns were obtained by a lord sampler after termination of leaching process to determine electrical conductivity (EC), pH, and sodium adsorption ratio (SAR). All measurements were conducted according to standard methods outlined by Page (1982).

Results and discussion

Table 1 indicates some properties of Shura and Sabakh soil. It is obvious that most total soluble ions exist in the upper 30 cm of both soils, which exceeded 50% of total soluble ions in soil columns. Maximum amount of soluble sodium was recorded in top layer and reduced with depth in both soils which is in agreement with numerous studies (Black 1973). Soluble sodium in different soil layers ranged from 84 to 440 mmol/L and from 207 to 682 mmol/L in Shura and Sabakh soil, respectively. Leaching curve shown in Figure 1 indicate that leaching water equivalent to less than half pore volume was enough to remove most soluble salts from soil columns within approximately ten days of leaching. Data (Table 2) shows tremendous decline in sodium adsorption ratio (SAR) in both soils which may indicate that sodium hazard was markedly eliminated. Soil analysis (Table 3), however showed that high amount of sodium still existed in soil columns even the EC of percolated water dropped below 4 dS/m. Percent of total sodium not removed from soil columns throughout leaching process ranged from 25% to 53%, and from 20 to 46 for Shura and Sabakh soil, respectively. This may indicate that resalinization of reclaimed soils is imminent due to this high sodium content left in the soil.

The relatively high amount of Na remained after reclamation is in fact a significant factor in resalinization through equilibrium process between solid and liquid phases of soil especially under dry conditions at which upward movement is much higher than the downward movement. A recent study by Finlayson and Raid (2007) still confirm that resalinization can occur if there is upward movement of salts by capillary action from a high saline water table, and ignoring the soil status after leaching especially Na status. Moreover,

Table 1. Properties of the soils used in the study.

Property	Shura soil			Sabakh soil		
	0-30 cm	30-60 cm	60-80 cm	0-30 cm	30-60 cm	60-80 cm
Sand (g/kg)	124	94	134	120	74	257
Silt (g/kg)	365	474	354	238	350	180
Clay (g/kg)	508	438	512	552	577	563
Texture	Clay	Silty clay	Clay	Clay	Clay	Clay
pH (1:5)	7.25	7.82	7.78	7.05	7.32	7.36
EC1:5 (dS/m)	17.8	6.4	4.3	35	6.4	5.3
$CaCO_3$ (g/kg)	310	340	290	310	310	360
OM (g/kg)	8.6	7.2	6.3	9.5	7.9	7.7
CEC (cmol/kg)	20	21	23	25	23	20
Gypsum(cmol/kg)	0.07	0.03	0.01	0.2	0.05	0.07
Na (mmol/L)	440	66	84	682	231	207
SAR	38	15.4	17.7	51	20	18

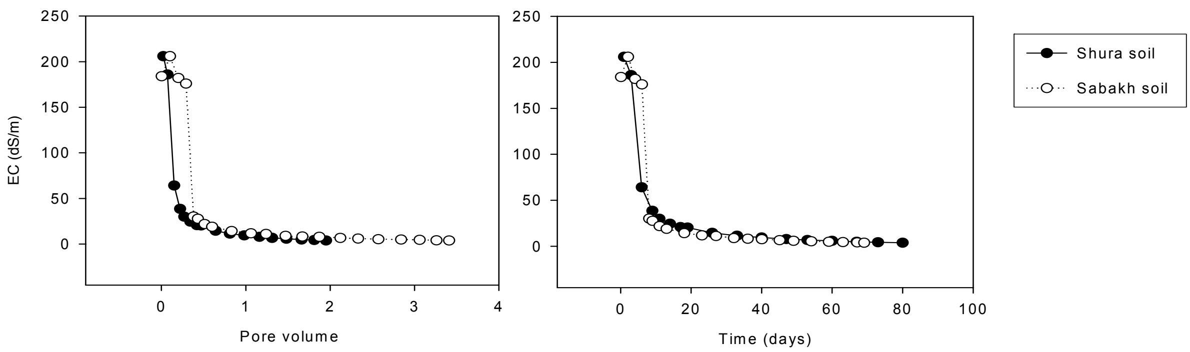

Figure 1. Effect of leaching process and leaching time on electrical conductivity for Shura and Sabakh soils.

Table 2. Effect of leaching process on reclamation parameters of soil on the basis of water extract (1:5).

Soil	Depth (cm)	EC (dS/m)		pH		SAR	
		Before Leaching	After Leacing	Before Leaching	After Leacing	Before Leaching	After Leacing
Shura	0-30	17.8	0.94	7.25	7.73	38	3.1
	30-60	6.4	1.24	7.82	8.10	24	1.7
	60-80	4.3	1.29	7.78	8.22	20	2.8
Sabakh	0-30	35.0	0.68	7.05	7.83	51	2.7
	30-60	6.4	1.5	7.32	7.76	20	2.1
	60-80	5.3	1.54	7.36	8.03	18	4.9

sodium salts are highly soluble in water and a high ratio of sodium in salt affected soils is soluble in soil solution (Black 1973) but this is in no contrast with our results of high sodium remaining along soil profile after leaching because this amount may include sodium salts occluded by soil particles, these soil particles are connected by very small pores and so they are not in direct contact with leaching water moved throughout soil. Sodium remained may also be as sodium salts mixed with another salts of calcium and magnesium which are less soluble, or even sodium salts occluded by clay minerals (Muhawish 1995). Analyzing leachates (Tables 4 and 5) confirm the existence of this amount of sodium ions along soil columns. This method of determination was considered by Panin (1962) to be more accurate to determine leaching efficiency than determination of salt amount in water extract of soil before and after leaching. Results of Table (4) and (5) also indicate that there is a cumulative percentage which valued 65% and 70 % (from initial total sodium content) in leachate of Shura and Sabakh soil respectively. This means that the remaining of total sodium is relatively equal to values obtained by analyzing soil for total Na. This relatively high level of Na remained after reclamation may be one of the main factors led to low productivity of recently reclaimed soils. According to these results total sodium content remained in soil is suggested to to be considered as a reclamation parameter.

Table 3. Effect of leaching process on total Na along columns of Shura and Sabakh soils.

Soil	Depth (cm)	Total Na before leaching (mg/kg)	Total Na after leaching (mg/kg)	Total Na remained as percent of initial (%)
Shura	0-10	19140	4828	25
	10-20	10480	5012	47
	20-30	11980	5725	47
	30-60	10320	4783	46
	60-80	9160	4863	53
Sabakh	0-10	26520	5206	20
	10-20	13560	4572	34
	20-30	15280	5059	33
	30-60	9290	4276	46
	60-80	11320	4968	44

Table 4. Amount of sodium accumulated in leachate of Shura soil

Sample No.	Sample volume (L)	Sodium Conc. In sample (g/L)	Sodium wt. in sample (g)	Percent of total sodium (%)	Cummulative percent (%)
1	7.56	53.61	405.3	49.3	49.3
2	5.17	7.95	41.1	5.0	54.3
3	5.76	4.63	26.67	3.24	57.5
4	4.02	4.6	18.49	2.25	59.79
5	5.42	2.24	12.14	1.48	61.27
6	7.34	1.07	7.85	0.96	62.23
7	27.36	0.85	23.26	2.83	65.06
8	7.92	0.57	4.51	0.55	65.61

Note 1: Percent of total sodium was calculated as follow; $\text{Percent of total sodium in column} = \frac{\text{Wt. of sodium for each sample}}{\text{Total wt. of sodium per column}}$ X 100

Note 2: Total wt. of sodium per column = 821.3 g

Table 5. Amount of sodium accumulated in leachate of Sabakh soil

Sample No.	Sample volume (L)	Sodium Conc. In sample (g/L)	Sodium wt. in sample (g)	Percent of total sodium (%)	Cummulative percent (%)
1	5.2	36.2	188.2	19.8	19.8
2	6.06	42.7	258.8	27.0	46.8
3	5.88	24.3	142.9	15.0	61.8
4	4.87	2.1	10.24	1.0	62.8
5	3.55	2.4	8.52	0.8	63.6
6	21.39	1.89	40.23	4.0	67.6
7	2.7	1.1	2.97	0.3	67.9
8	22.64	0.82	18.56	1.9	69.8
9	17.63	0.47	8.86	0.9	70.7

Note: Percent of total sodium was calculated as for Table 4.

Conclusions

It was concluded that the traditional parameter (EC of the drain water) seems to be inappropriate to determine neither extent of leaching process nor its completion because EC of the leachate never reflect amount of salts remained along soil profile. This study confirmed that approximately 40 % of total sodium initially present in salt-affected soils remained after termination of leaching process according to EC parameter. Therefore, total sodium remained in soil is suggested to be used as an effective index for reclamation process.

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Response of soil carbon pools to plant diversity in semi-natural grasslands of different land-use history

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Abstract
Within the scope of the BIOLOG-DIVA project, funded by the German Federal Ministry of Education and Research (BMBF), we analysed the effects of plant diversity on soil carbon pools as well as on biotic processes within the soil food web. It is a large interdisciplinary research project focusing on the relationship between biodiversity and ecosystem functioning in grassland ecosystems. The results of our study indicate that carbon pools and microbial community composition are influenced by the quantity of plant litter input. Processes of C accumulation are closely linked to vegetation quality and composition which have to be considered in future research.

Key Words
Land-use history, carbon pools, plant diversity, microbial community.

Introduction
Environmental changes and global climate changes have reduced biodiversity due to nutrient enrichment and increasing atmospheric CO_2 concentration (Smith *et al.* 2008; Walker *et al.* 2004). Biodiversity changes can alter the functioning of ecosystems such as biochemical cycles or the stability of ecosystems. This topic has raised numerous concerns including the fact that ecosystem functions such as carbon (C) accumulation may respond either positively or negatively (Naeem *et al.* 1996; Tilman *et al,.* 1996). Plant diversity can promote the accumulation of C in grasslands by enhancing productivity (Tilman *et al.* 2006); therefore less diverse ecosystems could weaken the ability of soils to accumulate C. The C reservoir of soils is about three times higher than the atmospheric C pool (Jobbagy and Jackson 2000) and steadily increasing atmospheric CO_2 concentrations have encouraged much interest in the long-term C capture potential of soils. It is important to understand how changes in plant species number and community composition in combination with changing management practices may influence rates of C accumulation (Fornara and Tilman, 2008). So far less information is available of biodiversity effects on soil C pools. Nevertheless, Fornara and Tilman (2008) found strong effects of plant functional composition on rates of soil C and N accumulation in a grassland biodiversity experiment. Steinbeiss *et al.* (2008) focused on soil organic C in the short term as a possible indicator for future soil C development. Higher diversity and plant functional traits increased the retention of C in the soil. This indicates that more diverse ecosystems could increase the C capture potential from the atmosphere to the soil. A changing management regime could therefore strengthen the ability of increasing the diversification of grassland ecosystems as this offers a possibility to reduce and store atmospheric CO_2 in soil.

The presented work is part of an interdisciplinary research project within the BIOLOG-Europe Programme (Biodiversity and Global Change), called DIVA. The project is funded by the Federal Ministry of Education and Research (BMBF) evaluating: “The relationship between Biodiversity and Ecosystem Functioning in Grassland Ecosystems”. It is a collaborative research effort of the Helmholtz Centre for Environmental Research (UFZ), Friedrich-Schiller-University and the Max-Planck-Institute for Biogeochemistry, both in Jena, and the Office for Ecological Studies in Bayreuth. Extensively managed meadows of different history in management intensity in the Thuringian and the Franconian Forest in Central Germany are used as model ecosystems to investigate the relationship between genetic/ phenotypic diversity and ecosystem processes such as C and nitrogen fluxes. The main objective of the presented work is the quantification of the four nodal points: input, transformation, accumulation, and loss of C in and from soils. We try to understand the biological and biochemical mechanisms behind these processes in relation to plant diversity and land use intensity in semi-natural grasslands.

Methods

Study site description

The study is conducted in semi-natural grasslands in Central Germany. All sites are located within an area of 20 by 40 km and are similar in elevation above sea level and exposition (Figure 1). The soil is a Stagnic Cambisol (siltic) and the bedrock material consists mainly of schist and greywacke. Average annual precipitation is above 950 mm with a slight summer maximum and the average temperature is 5.0 °C. All experimental plots are extensively managed, (no fertilization and grazing for the past 25 years) and mown twice a year in early July and September. The experimental plots represent a gradient in plant species diversity. They can be classified into two meadow types differing in species richness and biomass production: *1st* mountain hay meadows (high plant diversity but unproductive) and *2nd* rich meadows (low plant diversity but highly productive) (Table 1).

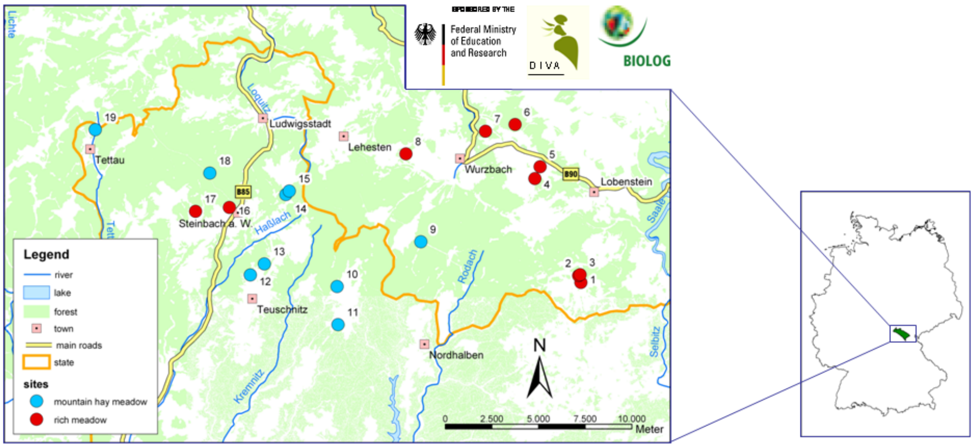

Figure 1. Overview of all experimental plots in the Thuringian and Franconian Forest in Central Germany.

Table 1. Study site and general plot description including soil texture and soil chemical properties. "Bayerische Landesanstalt für Wald und Forstwirtschaft in Freising" provided the climatic data. All values are given as means ± SE.

	Mountain hay meadows	Rich meadows
Average annual temperature [°C]	6.43	
Elevation (m)	645 (± 14.04)	638 (± 2.06)
Total annual precipitation [mm]	1346.4	
Species richness [N]	19 (± 2)	16 (± 2)
Aboveground biomass [g_{dm}/m^2]	134 (± 11.05)	218 (± 28.36)
Soil type	Stagnic cambisol	
pH (KCl)	4.13 (± 0.05)	4.88 (± 0.11)
SOC [%]	4.14 (± 0.35)	3.52 (± 0.26)
TN [%]	0.34 (± 0.03)	0.31 (± 0.02)
Clay [%]	27.03 (± 1.14)	23.91 (± 0.66)
Silt [%]	48.89 (± 1.49)	50.28 (± 1.19)
Sand [%]	24.08 (± 1.85)	26.93 (± 1.20)

Soil and vegetation analysis

Soil samples were taken in June 2008 from mountain hay meadows and rich meadows and separated into three depths (0-10 cm, 10-20 cm, 20-30 cm). The soil was sieved to < 2 mm, handpicked free from visible plant residues and stones. Soil samples for microbial parameters were frozen after sampling at -20 °C. Aboveground biomass was harvested at peak standing biomass and in accordance to the local management practices the aboveground biomass was removed from the plots. Soil organic carbon (SOC) and total nitrogen content of bulk soil samples were determined by combustion in a C/H/N analyser (Vario El III, Elementar-Hanau).

Particle size, density fractionation and C stocks

Physical fractionation methods include the separation according to particle size and density of primary organo-mineral complexes after disrupting the soil (von Lützow *et al.* 2007). The procedure follows the protocol of Shaymukhametov (1985) and was modified by Schulz (2004). For the fractionation procedure two replicates of 20 g (DM) bulk soil are used. Plant residues are separated by flotation with deionised water.

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The fractionation procedure consists of two steps: (i) the separation of SOM associated to an easily dispersible clay fraction (particles < 2 µm) by applying ultrasonic energy to a soil/water suspension; by different centrifugation speed and time the clay fraction (CF) was subdivided into a CF1: < 1 µm and CF2: 1 - 2 µm. (ii) After centrifugation the sediment was shaken with bromoform/ethanol mixtures of a density of 2 g/cm^3 and 1.8 g/cm^3, respectively, to stepwise separate two light fractions (LF); LF1: < 1.8 g/cm^3 and LF2: 1.8 - 2.0 g/cm^3. Under consideration of the bulk soil density and the stone mass content, SOC stocks (mass C/area) were calculated as described in Don *et al.* (2007).

Hot water extractable C and N
According to the method of Schulz and Hoffmann (2003), hot water extractions (HWE) were realized. Air dried soil samples were boiled in 50 ml deionised water or 1 h under a reflux and then membrane filtered. The plant material was boiled for 2 h under a reflux and the extracts were membrane filtered twice. Extracts of soil samples and plant material was analysed for the total C and N concentrations using an elemental analyser for liquid samples (Micro N/C and Multi N/C, Analytik Jena, Germany).

Substrate incubation experiment
The experimental units consisted of: soil by itself (Albic Luvisol, control) and soil amended with the respectively biomass (shoot- and root biomass). The soil was rewetted with distilled water and the finely powdered biomass was added to the soil on the basis of 500 mg C/kg. The soil was incubated in an automatic respirometer (RESPICOND, Nordgren 1988) for 30 days and CO_2 evolution was measured hourly.

PLFA and soil enzyme analysis
PLFAs will be extracted based on the method of Bligh and Dyer (1959) and Bausenwein *et al.* (2008). All enzyme activities (alkaline phosphatase, protease, β-glucosidase and xylanase) will be measured colorimetrically according to Schinner *et al.* (1993).

Results
It is accepted that the SOC dynamics represent an equilibrium between C input (e.g. primary productivity) and C output (e.g. decomposition) processes. These processes are driven by the interaction of abiotic and biotic factors. To better understand the SOC dynamics, the biotic linkages and the factors that regulate the transformation of C in managed grasslands we applied the following approaches:

Plant diversity effects on C pools and SOC stocks
Ecological relevant pools of organic matter were isolated through the combination of fractionation methods according to differences in particle size and specific density. The effects of plant diversity and land-use history on these pools will be presented.

Response of labile HWE SOM pool to plant diversity and its relation to the soil microbial community
We will extract a labile C pool through HWE and show how this pool is affected by plant species and litter quality. Additionally, the effects of this labile pool on phospholipid fatty acids and enzyme activities will be presented.

Litter quality as a key component of SOC pool formation
Decomposition of litter can be seen as a key process for the flow of energy and nutrients in terrestrial ecosystems and therefore we tested in a respiration experiment the decomposability of plant litter of plots differing in species richness.

Conclusions
From our results it can be concluded that decomposable but also stable or stabilized SOM pools should be considered for predicting the long-term C capture potential of soils for atmospheric CO_2. Future work of biodiversity effects on C accumulation in soils needs to consider the whole soil profile, since a huge amount of C is expected to be stored in deeper soil layers.
Details on actual results and on those of ongoing experiments will be presented at the conference.

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Response of soil microorganisms to land-use change in China, Ecuador and Germany

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Abstract
Within different ecosystems of China, Ecuador and Germany the effects of land-use change on i) nutrient turnover, ii) microbial biomass and iii) microbial community structure were assessed. On the Loess Plateau of China accelerated soil erosion, induced by land-use change from forest to agriculture, was the main process leading to soil degradation. Intensified soil erosion significantly decreased the contents of organic matter and microbial biomass in the soils. However, independent of erosion intensity N and C cycling rates were maintained in the bare plots. This might be explained by changes in the structure of the soil microbial community. In the mountain rainforest region of the South Ecuadorian Andes the initial increase in microbial biomass and activity after forest to pasture conversion by slash-and-burn slows down with increasing pasture age and/or increasing dominance of bracken fern and was associated with a significant shift in the phospholipid fatty acid (PLFA) fingerprint of the soil microbial community. A fast response of soil microorganisms to fertilization with urea was detected. At the agricultural study site in NE-Saxony, Germany, land-use effects were visible 6 years after starting the different management systems (intensive agricultural use, fallow) as indicated by the principle component analysis of PLFA data.

Key Words
Soil microbial community, PLFA, SOC mineralization, gross N mineralization, soil erosion, fertilization.

Introduction
Soil microorganisms are important drivers of nutrient cycling processes in soils. Land-use change as well as management practices are known to have impacts on soil microbial community structure and activity. However, extent and direction of land-use and management induced changes are highly variable and seem to depend strongly on the ecosystem considered. Some authors report shifts in soil microbial community structure within 2 years after land-use had changed (Hedlund 2002), others after 45 years (Buckley and Schmidt 2003). The interactions of microbial community dynamics and mineralization processes are complex and up to now not fully understood. For example, only little is known about how microbial communities mediate N cycling rates after disturbance of the soil ecosystem (Smithwick *et al.* 2005). Land-use induced disturbances of soil ecosystems like clear-cutting of forest, slash-and-burn practices and intensive agricultural use frequently occur worldwide often leading to soil degradation. Within the present study the response of soil microorganisms to land-use change in different ecosystems of China, Ecuador and Germany is compared. On the Loess Plateau of China accelerated soil erosion, induced by land-use change from forest to agriculture, is the main process leading to soil degradation. Soils with different degradation and rehabilitation status have been examined. In the mountain rainforest region of the South Ecuadorian Andes natural forests often have been converted to pastures by slash-and-burn. With advanced pasture age the pasture grass (*Setaria sphacelata*) is increasingly replaced by the tropical bracken (*Pteridium arachnoideum*) leading to the abandonment of this unproductive pastures (Beck *et al.* 2008). A sustainable management strategy for already existing pasture land in this mountain rainforest region of Southern Ecuador is one prerequisite to prevent further forest clearing for the establishment of new pastures. Hence, a pasture fertilization experiment was established where urea is used as N-fertilizer. In NE-Saxony, Germany, the effects of agricultural management practices on nitrogen dynamics and the structure of the soil microbial community have been considered depending on season.

Methods
Study sites
The study sites in China were located close to the "Fuxian Observatory for Soil Erosion and Eco-Environment" (Shaanxi Province), in the sole forest region remaining on the Loess Plateau. Annual mean air temperature and precipitation range between 6-10 °C and 600-700 mm. The forest is a 140 years old secondary stand. In 1989 within the forest runoff plots have been established on a hillslope to quantify soil erosion and the associated nutrient loss from the soils after clear-cutting as described in Zheng *et al.* (2005).

A land-use gradient from bare soils without vegetation (Bare), soils under agricultural management (Arable), soils with six year old natural successional vegetation (Succession) and soils under a 140 year old secondary forest (Forest) have been investigated. Additionally, a sequence of increasing erosion intensity along a hill from top to down slope was included on bare and forest sites (gradient: sheet erosion, rill erosion, gully top, gully). Soil samples were taken from 0-20cm depth. Soil type is a Calcaric Regosol developed on loess (Hamer *et al.* 2009a) with a soil pH(H_2O) between 8.5 and 9.0 (Table 1).

The study sites in Ecuador were located close to the "Estacion Científica San Francisco", about halfway between the provincial capitals Loja and Zamora, in the Cordillera Real, an eastern range of the South Ecuadorian Andes at about 2000 m above sea level. The mean annual air temperature is 15.3°C with an average annual rainfall of 2176 mm (Bendix *et al.* 2006). Soil samples were taken from 0-5cm depth of an active and an abandoned pasture site (Pasture, Abandoned Pasture). At both sites the soil type is a Cambisol and soil pH(H_2O) is acidic (5.4-5.6, Table 1). Using ^{14}C- and ^{15}N-labelled urea the effects of urea fertilization on soil organic matter mineralization and microbial community structure were investigated (Hamer *et al.* 2009b).

The study sites in Germany were located in Kreinitz (NE-Saxony) on a former arable site, which had been under fallow between 1996 and 1999. In 1999 the area was ploughed and divided into 12 plots of 50 m * 18 m size. Six out of these 12 plots were randomly chosen for intensive agricultural use (Intensive) and six plots were set aside and left to develop under natural succession vegetation (Fallow). Soil samples were taken from 0-10cm depth in June and September 2005. The soil type is a Cambisol developed on a loamy sand loess overlying a sand-gravel deposit with a soil pH(H_2O) between 5.3 and 5.6 (Table 1). Annual mean air temperature and precipitation vary between 8.4°C to 9.8°C and 550 mm to 600 mm, respectively (Hamer *et al.* 2008, Hamer and Makeschin 2009).

Table 1. Soil pH, soil organic carbon (SOC) content, C/N ratio, total amount of phospholipid fatty acids ($PLFA_{tot}$), mineralization of SOC during 14days of incubation and gross N mineralization rate in soils from China, Ecuador and Germany under different land-use (mean values; nd not determined).

	pH(H_2O)	SOC (g/kg)	C/N	$PLFA_{tot}$ (nmol/g)	SOC mineralization (%)	Gross N mineralization (mg/kg/d)
		China[1]				
Bare Sheet Erosion	8.7	7.5	10.2	11.8	1.1	0.2
Bare Rill Erosion	8.8	6.9	8.8	7.2	1.2	nd
Bare Gully Top	9.0	3.1	8.5	2.3	3.8	nd
Bare Gully	8.9	5.3	8.9	6.0	1.4	0.2
Forest	8.5	20.5	10.9	66.9	1.3	1.1
Forest Gully	8.6	19.8	11.1	44.0	1.0	1.6
Succession	8.8	11.0	9.8	22.6	1.2	0.5
Arable	8.8	5.8	8.6	23.1	2.2	0.5
		Ecuador[2]				
Pasture	5.4	122.3	12.5	322.0	1.2	12.1
Abandoned Pasture	5.6	78.1	15.7	124.0	0.8	0.4
		Germany[3]				
Intensive Agriculture (June)	5.3	8.2	11.6	16.1	0.5	1.1
Fallow (June)	5.6	9.2	11.5	12.0	0.6	1.7
Intensive Agriculture (September)	5.2	7.9	11.4	17.8	1.0	1.1
Fallow (September)	5.6	9.2	11.9	18.9	1.0	1.4

[1]0-20cm, n = 3; [2]0-5cm, n = 6; [3]0-10cm, n = 6

Microbial biomass and community structure
Microbial biomass carbon and nitrogen (MBC, MBN) were determined with the chloroform-fumigation extraction method. The structure of the soil microbial community was assessed using phospholipid fatty acid analysis (PLFA) as described in Hamer *et al.* (2009a,b).

Microbial activity
Mineralization of soil organic carbon (SOC) was determined during 14 days of incubation in the dark at 22°C. During the incubation the CO_2 produced was absorbed in 0.05*M* NaOH solution and quantified by titration. Gross rates of N mineralization were determined using the ^{15}N isotope pool dilution method as described in Hamer *et al.* (2009a,b).

Results and Discussion

At all three investigation sites in China, Ecuador and Germany the respective land-use change did not only affect the activity of soil microorganisms (Table 1) but also their community structure (Figure 1). As observed in laboratory incubations these changes are at least partly triggered by litter quality (Potthast *et al.* 2010) and fertilization regime (Hamer *et al.* 2009b). In the Ecuadorian pasture soils fertilization with urea induced a shift in the microbial community in both examined soils into the same direction: towards a higher relative abundance of PLFA marker characteristic of Gram negative bacteria and fungi (Hamer *et al.* 2009b). Also in the German agricultural soils fertilization seems to be important. However, there seasonal differences between samples taken in June and September were more pronounced (Hamer *et al.* 2008, Hamer and Makeschin 2009). As indicated by the principal component analysis of all PLFA data, there is a clear separation of soil microbial communities of the German and Chinese arable soils along principal component 1 from the other Chinese soils and the Ecuadorian soils with different land-use (Figure 1). The second principal component separates the Ecuadorian soils from the Chinese soils with one exception. The soil taken at the gully top at the bare site in China, a site without vegetation since 10 years, showed a PLFA fingerprint comparable to those of the Ecuadorian soils (Figure 1) and also had the highest mineralization of SOC (Table 1). Intensified soil erosion significantly decreased the contents of organic matter and microbial biomass in the Chinese soils, leading to the lowest SOC and $PLFA_{tot}$ contents observed among all sites investigated (Table 1). However, independent of the intensity of erosion N and C cycling rates were maintained in the bare plots. This might be explained by changes in the structure of the soil microbial community. Gross N mineralization and gross NH_4 consumption rates were significantly highest in forest soils. Within the forest, after 140 years, nutrient contents, microbial activity parameters and the soil microbial community structure were similar, independent of former erosion intensity. Thus, these parameters developed into the same direction during 140 years of secondary forest growth. Even after six years of natural succession a partial re-establishment of soil properties toward forest conditions was detected (Hamer *et al.* 2009a).

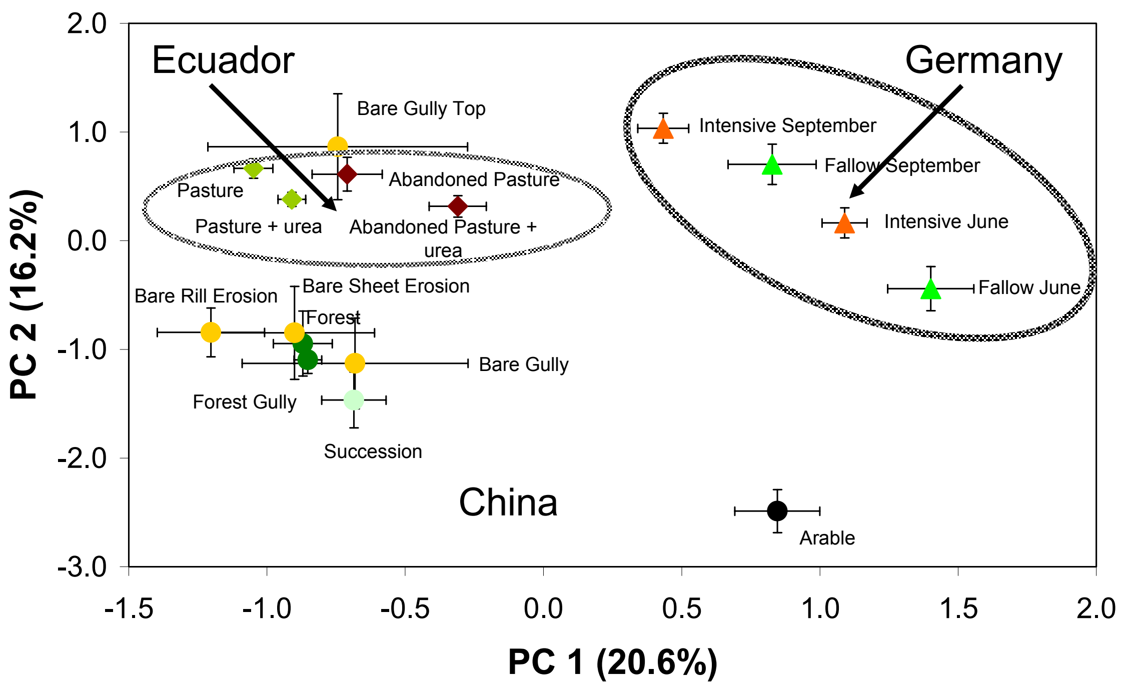

Figure 1. Principal Component (PC) analysis of PLFA data from soils of (◆) Ecuador (pasture and abandoned pasture without and with urea fertilization), of (●) China (bare, forest, succession and arable land with different erosion intensity) and of (▲) Germany (intensive agricultural use and fallow land in June and September) (mean values, bars represent standard error, n=6 for Ecuador and Germany, n=3 for China).

Conclusions

In the different ecosystems considered in China, Ecuador and Germany distinct microbial communities developed as indicated by their PLFA fingerprints. At all three sites land-use change significantly affected the soil microbial community structure and the microbial activity. Six years after land-use change these effects were detectable at the latest as can be seen at the German as well as Chinese sites. They might be detectable even more early. However, here further sites of different age have to be included. In laboratory incubation experiments it was obvious that soil microorganisms reacted immediately to fertilization. A changed microbial community structure was detected one month after urea addition to the Ecuadorian pasture soils.

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Soil attributes along an agricultural-forested gradient in a riparian zone

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Abstract
This study was carried out from January 2008 to December 2008 and aimed to evaluate modifications caused by the recovery of riparian vegetation on soil attributes. Transactions were defined in two margins of a dam, in order to represent continuous types of occupation (sugarcane-recent forest-old forest). Topsoil (0-0.2 m) and subsoil (0.2-0.4 m) samples were collected for evaluation of soil quality through chemical [pH, cation exchange capacity (CEC), and organic matter (OM), P, Ca and Al contents], physical (total porosity and apparent density) and microbiologic (basal respiration rate) attributes. Physical attributes were not affected for the soil use type and for vegetation cover. Significant differences ($p<0.05$) were observed for CEC, OM content and microorganisms activity according to soil use. CEC and OM content increased from the sugarcane plantation to the old forest, suggesting that the input of vegetal biomass through the recovered forest contributed to the improvement of chemical soil quality. Microbiological activity also indicated that the period of replacement of sugarcane plantation for recent forest was not still enough to cause changes in biological soil attributes. There was increase of the soil quality with the decrease of the use intensity and with the recovery of the natural vegetation.

Key Words
Rehabilitation of degraded areas, soil fertility, soil attributes, riparian forest.

Introduction
The concern and interest to preserve natural ecosystems and restore degraded ecosystems have been growing both in science and in public opinion. Native forests, represented by different biomes, are important ecosystems that have been exploited in an unsustainable way. The removal of natural vegetation and the establishment of crops, combined with inadequate management practices, cause disturbances in the interaction soil-plant by changes in the chemical, physical and microbiological soil attributes, limiting the use for agriculture and resulting an environment more susceptible to degradation (Centurion *et al*. 2004). This condition is aggravated when the surrounding agroecosystems affect natural forests, especially in riparian zones, since these plant communities provide the proper flow and/or drainage of water in watersheds, protect the riparian zone, mitigate the effects of excess of nutrients and of xenobiotic molecules, retain sediment, and prevent siltation of water bodies (Lima 1989). In the State of São Paulo, Brazil, there are one million ha of riparian areas that need to be recovered and reforested. In the season 2008/2009, approximately 4.43 million ha in the State of São Paulo were cultivated with sugarcane, which represent an important agroecosystem surrounding forest areas. The objective of this study was to evaluate changes in the chemical, physical and microbiological soil attributes promoted by restoration of native riparian vegetation and compare them with an agroecosystem intensively disturbed and cultivated with sugarcane.

Methods
Study areas
The study was conducted in two tracts of riparian ecosystems of 2 ha each, composed of two populations of semideciduous mesophytic forest, and an agroecosystem cultivated with sugarcane (SC), which concentrically bordering the dam of Santa Lucia sugar mill, located in the State of São Paulo, Brazil (22º18'00"S and 47º23'03" W; 611 m). The area has been cultivated with sugarcane during 30 years with conventional tillage and residue burning at harvest. The forested areas differ in age [a 9-year-old recent forest (RF) and a 18-year-old old forest (OF)], and with regard the soil type [Typic Hapludox (TH) and Arenic Hapludult (AH)], constituting six treatments (Figure 1).

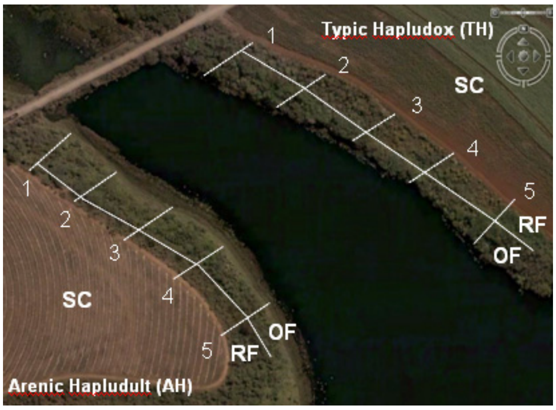

Figure 1. Representation of the transects in the experimental area (OF - old forest; RF - recent forest; SC – sugarcane) (Google Earth).

The climate is CWA (Köppen), i.e., mesothermal with hot and rainy summers and cold and dry winters. The average annual temperature is 21.4 º C and annual rainfall is 1448.8 mm (Brasil 1992).

Experimental characterization
For soil chemical and physical characterization, topsoil (0-0.2 m) and subsoil (0.2-0.4 m) samples were collected along five transects (Camargo *et al.* 1986; Raij *et al.* 2001) (Figure 1). To evaluate the evolution of soil quality by restoring riparian vegetation, the following attributes were considered: chemical - pH, cation exchange capacity (CEC), and organic matter (OM), phosphorus (P), calcium (Ca) and aluminum (Al) contents; physical – texture, bulk density (Da), total porosity (Pt), water retention capacity and field moisture measured *in situ* by time-domain reflectometer (TDR). Soil samples (0-0.1m) were collected along the transects 2, 3 and 4 for microbiological characterization, through basal respiration rate (Anderson and Domsh 1978). Results were submitted to analysis of variance and differences between means determined by Tukey test at 5% probability.

Results

Chemical attributes
Soil chemical characteristics are modified with the removal of natural vegetation and cultivation, especially in the topsoil. In general, a decrease of soil pH with the cultivation time is observed (Cerri 1986). However, in both soils (TH and AH), the pH of topsoil was higher in OF than in other environments. In the area planted with sugarcane, soil pH was higher in subsoil, which identified the likely acidifying effects action of fertilizers in topsoil and the deepening of the corrective effect of lime in the subsoil. With the establishment of natural vegetation, Ca levels increased in both depths of the TH. Higher Ca concentrations were observed in the cultivated area on AH, probably due to the recent liming. Low levels of Al were observed in TH, but in RF area of AH, Al contents were close to 10 $mmol_c/dm^3$, that is, a serious limiting factor for plant development. The concentration of P was higher in the area cultivated with sugarcane, especially in AH (Table 1).

Table 1. Selected chemical attributes of topsoil and subsoil samples of Typic Hapludox (TH) and Arenic Hapludult (AH) under different soil uses.

Depth(m)	Area	P	OM	pH	K	Ca	Mg	H+Al	Al	SB	CEC	V	S	B	Cu	Fe	Mn	Zn
		mg dm^{-3}	g dm^{-3}	$CaCl_2$	------------------- $mmol_c$ dm^{-3} -------------------							%	------------------- mg dm^{-3} -------------------					
								Typic Hapludox (TH)										
0-0.2	OF	6.0	33.8	5.2	3.2	35.0	13.2	35.0	0.7	51.4	86.4	59.4	8.4	0.4	5.2	28.4	53.6	1.9
0-0.2	RF	12.4	25.2	5.0	2.2	25.0	8.2	42.2	1.5	35.4	77.6	46.3	8.2	0.2	4.4	17.0	49.4	1.1
0-0.2	SC	16.4	24.2	4.8	3.8	19.6	6.8	40.6	1.9	30.2	70.8	42.6	38.4	0.2	4.1	13.8	48.6	0.9
0.2-0.4	OF	3.6	25.6	5.0	2.3	26.0	10.4	37.0	1.0	38.7	75.7	50.8	5.8	0.3	4.8	23.6	53.8	1.1
0.2-0.4	RF	6.0	20.8	4.9	0.9	20.8	5.0	41.4	1.6	26.7	68.1	39.5	11.8	0.1	3.6	13.6	38.2	0.6
0.2-0.4	SC	5.0	18.8	5.0	1.3	22.0	6.4	37.6	0.9	29.7	67.3	44.1	27.0	0.1	3.1	10.4	22.2	0.4
								Arenic Hapludult (AH)										
0-0.2	OF	4.3	24.9	4.7	3.8	19.3	9.4	33.9	2.3	32.5	66.4	48.5	9.7	0.7	2.1	63.8	42.3	1.7
0-0.2	RF	5.0	16.3	4.6	3.2	16.6	5.4	35.1	5.6	25.2	60.3	40.9	15.1	0.4	2.1	42.8	20.9	0.5
0-0.2	SC	40.4	20.9	4.6	5.6	23.6	8.6	41.7	3.9	37.8	79.5	47.2	31.1	0.6	2.4	61.2	18.5	0.9
0.2-0.4	OF	1.3	11.1	4.6	1.9	16.3	6.2	33.3	2.7	24.4	57.7	40.7	9.0	0.3	1.5	30.9	34.2	0.5
0.2-0.4	RF	2.5	11.2	4.4	2.0	15.2	5.1	44.4	9.6	22.3	66.7	34.7	18.7	0.5	1.6	30.5	27.1	0.4
0.2-0.4	SC	28.0	15.2	4.7	4.9	24.1	9.0	41.9	2.5	38.0	79.9	47.1	27.6	0.5	2.5	47.9	15.6	0.6

V - level of base saturation

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In both soils, OF showed greater accumulation of OM, due to the higher input of biomass. Major changes in OM content were observed in the 0-0.2 m layers, especially in TH. When compared with the SC area, soil under OF showed increase of 10 g/dm^3 and 6 g/dm^3 at depths of 0-0.2 m and 0.2-0.4 m, respectively. Less evident variations were observed in the AH, but in cultivated area and at depth of 0.2-0.4 m was recorded greater accumulation of OM, resulting from the conventional management practices adopted during several years, including the application of vinasse, which reaches depths, especially in soils with low clay content (Maia and Ribeiro 2004). Because of their close dependence of soil organic matter, CEC showed variations similar to those observed for the OM. Samples collected from cultivated area on AH had a higher CTC, due to liming and vinasse addition. However, in the TH, CEC was significantly larger in the forest, with severe decrease in the cultivated system. Clearly, OM content and CEC were considered the chemical attributes that most reproduced the increase of soil quality with the restoration of natural vegetation on the TH.
In cultivated systems, was observed a decrease in the micronutrients contents, usually associated with the decrease of OM contents and with the continuous application of formulated fertilizers without these elements. Levels of micronutrients were higher in OF when compared to the levels of soils under RF and SC, but are still low.

Physical attributes
Generally, main physical changes due to intensive soil management are the reduction of Pt, especially the macroporosity, and the increase in Da (Araújo *et al*. 2004). Values for Da varied within the range expected for mineral soils (1.10-1.60 g/cm^3) and were below those reported as limiting or with potential to cause restrictions to root growth (1.70-1.80 g/cm^3). Results for Da and Pt showed differences between the TH and AH, attributed to the texture. No change of Pt and Da for different soil uses or vegetation cover was observed. Although no statistically significant differences in physical attributes was observed, the input and stabilization of organic material increased Pt and decreased Da of soils under forest, since in these environments there is no soil tillage such as agricultural systems.
Clay content of TH was 565 g/kg, which did not vary along the transects. In AH, clay content ranged from 245 g/kg, in SC, to 140 g/kg, in OF. Probably, the topsoil of AH was removed and transported from SC (higher elevation area) to OF (vicinity of the dam) by erosion due to it sandy topsoil.
Water retention at field capacity (θcc) was 12.9 g/g for AH and 27.4 g/g for TH. Soil moisture was measured in four seasons, with variation along the transects of 18.7% in AH and 24.1% in TH. Higher contents of OM and clay, and more exuberant vegetation on TH were considered factors related to increased water content.

Microbiological attributes
The microbial growth is greatly influenced by climatic variations, especially moisture and temperature, as well as the effects that these variations have on the vegetal cover (Cattelan and Vidor 1990). The annual average temperature was 21.8^o C and precipitation in the period of 1306.6 mm (Figure 2). The water balance characterized 2008 as an atypical year with water deficiency.

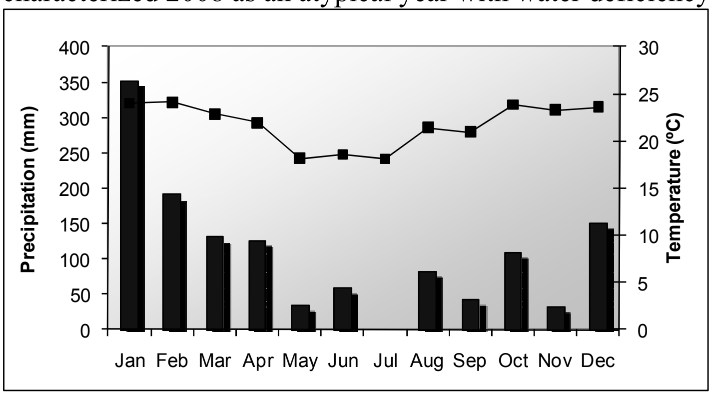

Figure 2. Precipitation and air temperature obtained for the period from jan/08 to Dec/2008.

Old forest on TH showed the highest levels of OM (33.8 g/dm^3 at 0-0.2 m). The OM is an important source of energy and nutrients for heterotrophic microorganisms (Alexander 1977) and increases the storage capacity of water in the soil (Hillel 1982), promoting microbial growth. Comparing the rate of CO_2 released by samples from the two soil types, microbial activity was lower in AH than in TH. Differences were attributed to the lower level of organic matter found in AH, even in OF area (24.9 g/dm^3).
Microbial community showed little fluctuation during the year, due to little climatic changes during the period (Figure 3).

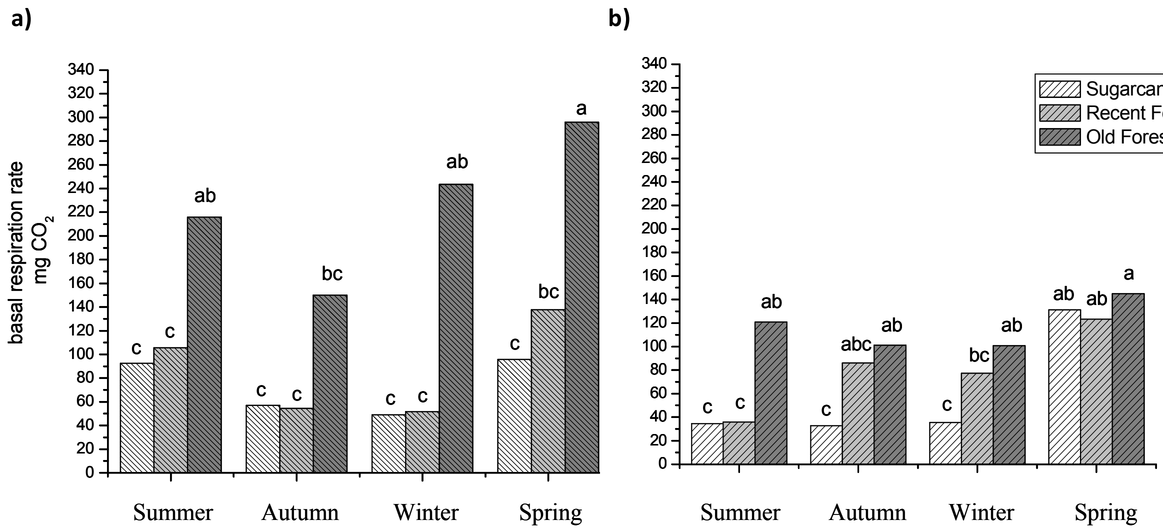

Figure 3. Soil bacteria (CFU/g dry soil) according to the different seasons, in (a) Typic Hapludox and (b) Arenic Hapludult; different letters indicate statistical differences according to Tukey test ($p<0.05$).

Old forest areas showed greater basal respiration rate, while in SC and RF systems, values were significantly lower. For both soils, the time to replace the culture of sugarcane by the most recent native vegetation was not sufficient to cause significant changes in biological properties of soil.

Conclusions

There was an increase of soil quality by reducing the intensity of use and the restoration of natural vegetation. The organic matter content, the cation exchange capacity, and microbial activity were the soil properties most sensitive to changes of soil management and reproduced more variations of soil quality with respect to soil use and vegetal cover.

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Soil quality benefits of break crops and/ or crop rotations-a review

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Abstract

Cropping systems in Western Australia (WA) are strongly dominated by cereals (wheat, barley and oaten hay) and are inextricably linked to soil quality and moisture supply. So a major research challenge is to devise cropping systems that improve soil health and maximise water-use efficiency. For a profitable and sustainable production system there is a clear need for 'break crop' to provide a pest, disease and weed break, conserve soil moisture; adding soil organic matter and nutrients and to help alleviate soil physical and structural problems. This review article demonstrates the role of 'break crops' and/or of 'crop rotations' in providing potential soil quality benefits for profitable and sustainable crop production systems.

Key Words

Break crops, crop rotation, soil quality.

Introduction

Most soils in WA were extremely deficient in N, and not capable of supporting profitable yields of wheat until the introduction of legumes into the rotation. The introduction of grain legume crops or legume-rich pastures provided N to a subsequent cereal or oilseed crops (Rowland *et al.* 1988, 1994). The use of lupins as a major rotation crop for sandplain soils coincided with the availability of herbicide options and reduced tillage. Lupins provided farmers with a profitable alternative to pasture legumes and gave them an option to control grass weeds using herbicides in the non-cereal phase of the rotation. An unknown part of the observed yield increase of wheat following pasture legumes was probably also due to a reduction in root diseases. More recently, the widespread use of canola in many rotations has added an extra management tool to maintain profits and for controlling grass weeds. However, the yield increases in the following wheat crop that were observed in eastern Australia (Angus *et al.* 1991) have been less evident in the west, possibly due to poorer soil fertility. The reason there is so much interest in crop rotations in WA (or elsewhere) at the moment is because there is a sense of belief that this will move the current farming system towards a more 'sustainable' farming system. With the monoculture (of wheat/barley/oaten hay) and the predictability of the current system, any variation in the cropping rotation would be of benefit.

Choice of break crops to grow in addition to cereals and the fertility building phase are crucial to both the agronomic and economic success of the rotations. There are four specific functions that a break crop may perform, namely - improvement in soil health; conservation of soil moisture; weed control and pest and/or disease control. Individual break crops may perform one or several of these functions. A good break crop is also expected to produce satisfactory yields, be of marketable quality, and produce an economic return for the farmer. The main principles of crop rotation design are described in Table 1.

The appropriate choice of crops within the rotation and their sequence are crucial if nutrient cycling within the farm system is to be optimised and losses minimised over the short and long term. Each crop species has slightly different characteristics, e.g., N demanding or N_2 fixing, shallow or deep rooting, amount and quality of crop residue returned. Theses characteristics along with exiting biotic and abiotic factors determine the ultimate suitability of a break crop in a given cropping system.

This review highlights the fact that much information currently existing in the scientific literature on role of break crops or crop rotations could be utilised to optimise certain management practices for profitable and sustainable crop production. The topics discussed in this review on soil quality are in the order of their importance to the West Australian cropping systems.

Soil nitrogen

One of the persistent nutrient management questions associated with the legume rotation is whether the N contribution from legume fixation is responsible for much, if not all, of the beneficial rotation effect. Many studies have shown that cereals derive both yield and N benefits from rotations with grain legumes compared with cereal monoculture (Kirkegaard *et al.* 2008). The yield advantage may be entirely due to N or to other factors, but more commonly a combination of both (Chalk 1998). Evans *et al*. (2001) estimated that average net N input from grain legumes to be 47 kg/ha N in south eastern Australia and 90 kg N/ha in south western Australia. Removal of the above ground pea residues, which contained less than 1% N, had no effect on residual N value. Stevenson and van Kessel (1996) found that 91% of the wheat yield benefit from a preceding pea crop came from reduced leaf disease and weed infestation, while only 9% was estimated to have derived directly from N. Benefits in N nutrition to wheat may also arise from break crops simply because the healthier root system is able to utilise existing soil N or applied N more efficiently (Cook 1990). Non-legume break crops may also differ significantly in the amounts of mineral N left in the profile. Kirkegaard *et al*. (1997) found that residual N remaining after a range of winter oilseeds was a key factor in determining subsequent wheat yields in the absence of disease. Linseed had a shallower rooting system, produced less biomass and left 30-50 kg/ha more N in the profile at harvest than canola or mustard. Accumulation of mineral N from break crop residues may also differ during the fallow period prior to cropping and this may not be simply related to C: N ratio of the residues (Kirkegaard *et al.* 1999).

Table 1. Rationale of crop rotation in farming systems.

Principle	Reasoning
Rotate deep and shallow rooting crops	Improve soil structure, aeration, water-holding capacity, and drainage
Alternate crops with large and small root biomass	High biomass crops increase the organic matter remaining in the soil for soil microbial and macrofaunal populations
Rotate N_2-fixing and N-demanding crops	Attempt to meet farm's N demands from within the system
Alternate weed-susceptible and weed suppressing crops; rotating varieties of same crop e.g. EGA Eagle Rock wheat vs Clearfield wheat, Mandelup lupin *vs* Jenabilup lupin, TT canola *vs* IT canola or conventional canola	Interrupt weed life cycle to reduce populations; decreasing likelihood of developing herbicide resistance in weeds by using different herbicides for respective variety and their varied mode of action
Grow crops with different pest and disease susceptibilities	Break pest and disease life cycles, reduce host plant presence in rotation
Grow catch crops, green manures, and undersow crops	Maintain soil cover to protect for erosion and leaching
Balance forage and cash crops	To make rotation economically as well as ecologically viable
Fallow	Soil water storage, weed control

Soil organic matter

Soil organic matter (SOM) is a key indicator of quality as it influences biological activity, serves as a nutrient reservoir, and impacts soil aggregation. Heenan *et al.* (2004) from a long term rotational study in Wagga Wagga NSW reported that stubble retention in legume-wheat rotation maintained higher levels of SOC than stubble burning. The effects of management treatment on soil total N were similar to effects on SOC. Where change in SOC and TN occurred, there was no evidence that equilibrium had been reached, although a change in slope had occurred in many treatments. Masri and Ryan (2006) from Cropping System Productivity trial at ICARDA Syria reported that some rotations, e.g., medic (*Medicago sativa*) and vetch (*Vicia faba*), significantly increased soil organic matter (12.5–13.8 g/kg versus 10.9–11 g/kg for continuous wheat and wheat/fallow).

Phosphorus

Phosphorus (P) is a major limiting nutrient for crop production on many Australian soils due to high P fixation and low levels of plant-available soil P. Only 10-20% of the applied P is utilised by crops in the year of application and subsequent usage of the residual P rarely exceeds 50% (Bolland and Gilkes 1998). Fertiliser P reacts with soil constituents and is readily 'fixed' as adsorbed P, sparingly soluble P-precipitates (Al-P, Fe-P or Ca-P) or converted to organic forms that are largely unavailable to most crop plants. Thus a major challenge is to find ways to improve the P use-efficiency of high P fixing soils. One approach that has potential benefits on P availability is the incorporation of P-mobilising species into the cropping system (Horst *et al.* 2001). Several legume crops can mobilise soil and fertiliser P through the exudation of organic-acid anions such as citrate and malate and other compounds from their roots from their roots eg, chickpea (Veneklaas *et al.* 2003), and white lupin (Keerthisinghe *et al.* 1998). This mechanism enables some of these species to acquire P from soil sources that are not readily available to non-secreting crops. A number of studies have reported improved growth and P nutrition of less P-efficient crops following organic-anion exuding legumes (Hocking and Randall 2001). Despite being a promising approach to improve the P-use efficiency of cropping systems, however little is known about the conditions (soil type, plant species) and mechanisms governing these benefits.

Soil structure and physical properties

In WA soil physical and structural problems have become wide spread particularly in certain soil types such as earthy sands and sandy duplex subsoils. Some break crops can be used to help alleviate such problems, either because of the nature of the break crop itself or as a result of cultivation methods used during production. The roots and residues of break crops may influence several aspects of soil structure through exudation or release of stabilising or destabilising substances in the rhizosphere, root and associated hyphal enmeshment or fragmentation, and the production of stable biopores. Reeves *et al*. (1984) in a south in a south Australian study reported that differences in soil water-stable aggregates and bulk densities following wheat and lupin crops were small and inconsistent. Chan and Heenan (1996) reported soil following canola and lupin was more porous, had lower soil strength and had stronger, more stable aggregates than soil after peas or barley, and the improvements related to the impacts of roots on soil aggregate formation and macro-pore creation. Interestingly, both lupin and canola are non-AMF (arbuscular mycorrhizal fungi) hosts so that the improvements in aggregate stability following those species could not be explained by glomalin production by the associated AMF as has been recently demonstrated for other crops by Wright and Andersen (2000). Cresswell and Kirkegaard (1995) reviewed the evidence for improvements in subsoil structure by break crops and concluded that the effects were either small, not evident, or could not be adequately distinguished from additional influences of break crops such as reduction in soil-borne diseases.

Conclusion

Benefits of break crops in rotation for crop productivity have been identified, but processes and mechanisms responsible for those benefits need to be better understood particularly under WA growing environments. There is serious lack of published on data under WA growing conditions. Most of the work reported in this review has been done elsewhere. Long-term trials are needed if reliable conclusions are to be reached on the performance of break crops in different WA climatic zones and soil types. This is a critical area for basic and applied research. No one ‘crop rotation’ will be the ‘right’ rotation, as the goals trying to be reached may vary and hence one situation may require one rotation and this may be completely unsuitable in another situation.

Dynamic Crop Sequence (DCS) trial in Katanning and Wongan Hills in WA started recently is the first step in the right direction to understand the role of break crops in WA cropping systems. A dynamic cropping system represents a long-term strategy of annual crop sequencing that optimises crop and soil use options to attain production, economic, and resource conservation goals by using sound ecological management principles. These trials look at quantifying the effects of crop residues on the following crop in the sequence with respect to water use, disease flow, stability of yield, economics and measures of soil health from a total of 100 treatment combinations generated from wheat (with and without Jockey fungicide seed treatment), barley, oats (grain and hay), field pea, lupin, canola, green manure and fallow. This is holistic approach in management of soil moisture, weeds, diseases and nutrients through use of break crops in rotation and suitable agronomic practices. The end objective of trials is to generate information about which crop is best to sow in what situation and how to combine crop sequences to maximise opportunities. This agronomic research will also be backed by an economic analysis to give farmers more confidence in the decisions they make about their cropping program in WA or elsewhere with similar growing conditions.

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Soil quality in a semi-arid Mediterranean soil as affected by tillage system and residue burning

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Abstract

We used principal component analysis (PCA) of soil physical, chemical and biological attributes to evaluate differences induced by three different tillage systems (conventional tillage (CT), reduced tillage (RT) and no-tillage (NT)), and stubble burning under NT (NTSB) in the topsoil of a carbonate-rich soil in semi-arid Mediterranean Spain. PCA resulted in three factors with eigenvalue >1, grouping parameters related to organic matter quality, physical status and organic N, respectively. These factors explained 82.71% of the variance. Scores for the different treatments grouped NTSB and CT for organic matter quality, NTSB with NT for soil physical status, and identified NTSB as the most different treatment for N parameters. Burning crop residues under NT affects the quality of organic matter and the N cycle more than the physical quality of the soil, which was better under NT and NTSB. The impact of stubble burning on the N cycle appears as a promising field of research.

Key Words

Soil quality, soil enzymes, stubble burning, semi-arid land, carbonate-rich soils.

Introduction

Many soils in semi-arid Mediterranean Spain are poor in organic matter. Intense agricultural management has resulted in poor soil quality (SQ) in many areas (Andrade 1998). Tillage and management of residues have been seen to modify SQ (Bescansa *et al.* 2006, Moreno *et al.* 2009) in these soils. For instance, no-till (NT) leads to greater organic matter contents and better physical condition (Fernández-Ugalde *et al.* 2009). Stubble burning is a controversial practice. Despite of its known environmental disadvantages, it is still practiced in many areas worldwide. It is usually justified as an efficient way of fighting pests and diseases. It is also useful to eliminate the excess of crop residues in some cases. When stubble is burnt shortly before seeding, and with low-intensity fires, it has been observed to alter the quality of organic matter (Virto *et al.* 2007) in semi-arid NT soils.

Soil quality, which has been defined as the capacity of a soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality, and promote plant and animal health (Doran and Parkin 1994), must be assessed accounting for both inherent and dynamic soil properties and processes and must be holistic (Karlen *et al.* 2003). SQ evaluation must therefore include the study of soil physical, chemical and biological properties (Doran and Parkin 1994). Best SQ indicators are those parameters which have greatest sensitivity to changes in soil function (Andrews *et al.* 2004).
The objective of this study was the evaluation of SQ in relation to different tillage practices and residue management, including stubble burning under NT on a carbonate-rich soil in semi-arid Mediterranean Spain. We considered a set of physical, chemical and biological properties, and compared their sensitiveness and their response to management practices. Then we run a principal component analysis (PCA) in order to evaluate the main differences found in the topsoil after 10 years in relation to management, and to identify the most sensitive indicators in relation to management in this soil.

Materials and Methods

Site description and experimental design

We studied the soil in an experimental field at Olite (42º27’19”N; 1º41’10”W; Alt.: 402 m a.s.l.) in Navarra (NE Spain). The soil is a Typic Calcixerept (Soil Taxonomy 2006). Climate is semi-arid Mediterranean with an annual average precipitation of 525 mm. Potential mean annual evapotranspiration is of 740 mm, and mean monthly annual temperature is 13.5 ºC.
The experiment was designed as a randomized complete block with four replicates (n=4). Four treatments were studied: conventional tillage (CT), reduced tillage (RT), NT with stubble standing (NT), and no-tillage with stubble burning (NTSB). Crop residue was incorporated into the arable layer in CT, which consisted of 0.25-m-deep primary tillage with a three-furrow mouldboard plough, then a smoothing pass with a float, and

sowing using a coulter-seeder. RT consisted of chisel ploughing (0.15 m deep) and secondary tillage and seeding as for CT. For NT and NTSB, a direct seeder was used, which opened the seed-row 3 to 5 cm deep. For NTSB stubble was burnt in October just before seeding. Barley was seeded each year in October-November (158 kg/ha). Fertilization was done according to crop needs and was equal among treatments.

Soil analysis

Soil samples were collected in the topsoil (0-5 cm). Composite samples were air-dried and ground to pass a 2-mm sieve, except for analysis of dehydrogenase activity. In this case, soils were sieved to 2 mm in fresh. After sieving, those soil portions for enzyme activity determination were stored at 4ºC until analysed in laboratory. For aggregate stability, a portion of air-dried samples were forced to pass a 8-mm sieve. A set of physical, chemical and biological properties were determined.

Particle size-distribution (PSD), bulk density (ρ_b), aggregate stability and penetration resistance (PR) were chosen as physical properties. No differences were observed in PSD among treatments. PSD was therefore excluded for PCA (see below). The core method was used to determine ρ_b. Aggregate stability was determined with homogenized samples, by placing 100 g of dry aggregates in the top of a column of sieves of 6.3, 4, 2, 1, 0.5, 0.25 mm openings and shaking the whole in a rotary movement at 60 strokes/min for 60 s in a Retsch VS 100 device (Retsch GmbH & Co. Haan, Germany). For wet aggregate stability, a constant shower-like flux (6 L/min) of distilled water was applied from the top of the same set of sieves while sieving (60 strokes/min, 60 s). Aggregate stability was expressed as the mean weight diameter after dry (MWD_d) and wet (MWD_w) sieving, from which the calculated the MWD_w-to-MWD_d, as proposed by Lehmann *et al.* (2001). Penetration resistance (PR) was measured on site to a depth of 60 cm using a field penetrometer (Rimik CP20, Agridy Rimik Pty Ltd, Toowoomba, QLD, Australia). Nine PR measurements per plot were recorded at 15 mm depth intervals with the soil uniformly wet, to avoid differences in moisture content among treatments. For statistical treatment, PR of the 0-5 cm depth was calculated as a weighed average.

Soil pH, electrical conductivity (EC), carbonates, P and K contents were analysed as chemical properties. None of these parameters showed statistical differences among treatments, and so that, they were excluded in PCA (see below).

Finally, total soil organic C (SOC), C in the form of particulate organic matter (POC), total N (N) and N in the particulate organic matter (PON), SOC mineralization (SOC_{min}) and the activity of five soil enzymes were included as biological properties. Due to the elevated concentration of carbonates in the soil (~35%), total soil organic C (SOC) and C in the form of particulate organic matter (POC) were determined by wet oxidation (Walkley-Black). Particulate organic matter (0.053-2 mm in equivalent diameter, Cambardella and Elliot (1992)) was isolated by dispersion and sieving of 10 g samples of air-dried soil, using a method adapted from Marriott and Wander (2006). Total N and PON were determined by Kjeldahl digestion. The C-to-N ratio of total and particulate organic matter was calculated from SOC, N, POC and PON data. SOC_{min} was estimated as the amount of CO_2 respired in a 14-days incubation at 55% field-capacity and 20 ºC. The activities of dehydrogenase and four enzymes involved in soil C (β-glucosidase), N (urease), P (acid phosphatase) and S (arylsulphatase) dynamics were determined according to a modification of Tabatabai (1994), as previously described (Rodríguez-Loinaz *et al.* 2008). Only differences in urease and acid phosphatase were observed among treatments. The activities of the other enzymes were excluded for PCA.

Statistics

Data were analysed using ANOVA (univariate linear model). Treatment means were compared using significant differences ($P<0.05$). As indicated above, only data corresponding to the variables showing significant differences among treatments were subjected to factor analysis using PCA. PCA allows grouping variables into statistical factors based on their correlation structure (Brejda *et al.* 2000). To eliminate the effect of different units of variables, factor analysis was done using the correlation matrix on the standardized values of the measured soil properties (Shukla *et al.* 2006). Using this correlation matrix, principal components (factors) with eigenvalues >1 were retained and subjected to varimax rotation with Kaiser to estimate the proportion of the variance of each soil variable explained by each selected factor (loadings). Factor scores for each sample point were calculated in order to evaluate the effects of the each treatment on the extracted factors. All statistical analyses were performed using SPSS 16.0.

Results and Discussion

Principal component analysis identified three factors (PCA-F1, PCA-F2 and PCA-F3) with eigenvalues >1 (Table 1). They explained 53.93, 19.80 and 8.97 % of the variance, respectively. PCA-F1 had the highest positive loadings from SOC, C/N ratio, CE, POC, SOC_{min} and phosphatase. This factor groups thus mostly parameters related to organic matter quality. PCA-F2 received the greatest loadings from ρ_b, PR and aggregate stability, grouping parameters related to the soil physical condition. Finally, PCA-F3 grouped the three studied parameters related to soil N (total N, PON and urease activity).

When the calculated scores of PCA-F1, PCA-F2 and PCA-F3 for each soil treatment were analyzed, significant differences were found. Scores of CT and NTSB were equal among them and smaller than those of RT and NT for PCA-F1, suggesting that the characteristics of organic matter in CT and NTSB were similar. Virto *et al.* (2007) showed indeed that although SOC stocks were similar under NT and NTSB in this soil, organic matter in NTSB was less easily mineralizable than in NT, most likely due to the inherent recalcitrance of partially burnt and charred plant residues. PCA confirms this trend. Interestingly, phosphatase activity was also grouped in PCA-F1, indicating that P dynamics is related to organic matter in this soil, and is different in NT depending on residues being burnt or not. In this respect, it has been reported that trigger molecules or promoters released by organic materials stimulate the production of hydrolytic enzymes such as acid phosphatase (Martens *et al.* 1992). More interestingly, Olander and Vitousek (2000) found that phosphatase synthesis was inhibited in the presence of readily available inorganic P in soil.

Table 1. Proportion of variance explained using varimax rotation for each of the factors with eigenvalue >1 (PCA-F1, PCA-F2 and PCA-F3) in the 0-5 cm depth. and scores of PCA-F1, PCA-F2 and PCA-F3 for CT (conventional tillage), RT (reduced tillage), NT (no till) and NTSB (NT with stubble burning).

	PCA-F1	**PCA-F2**	**PCA-F3**
Eigenvalue	**7.012**	**2.574**	**1.166**
SOC (mg C/g soil)	**0.791**	0.441	0.183
C/N ratio	**0.826**	0.452	-0.170
N (mg N/g soil)	0.276	0.215	**0.813**
CE (µS/cm)	**0.885**	-0.152	0.236
POC (mg POC/g soil)	**0.798**	0.385	0.172
PON (mg PON/g soil)	-0.061	0.363	**0.831**
SOC_{min} (mg CO_2-C/g soil)	**0.814**	-0.102	0.115
Bulk density (g/cm^3)	-0.053	**0.859**	0.287
MWD_w (mm)	0.273	**0.862**	0.280
MWD_w /MWD_d	0.282	**0.843**	0.244
PR (MPa)	0.049	**0.801**	0.505
Urease (µg N-NH_4^+/g soil/h)	0.284	0.465	**0.637**
Acid phosphatase (µg 4-NP/g soil /h)	**0.627**	0.311	0.592
Scores			
CT	-0.896 a	-0.568 a	-0.889 a
RT	0.368 b	-1.025 a	0.248 ab
NT	1.312 b	0.518 b	-0.196 ab
NTSB	-0.784 a	1.076 b	0.837 b

Within columns, values followed by different letters belong to different Duncan's homogeneous groups (P<0.05), for each factor. Bold figures indicate the factor receiving the greatest loading from each soil attribute.

Scores for PCA-F2 were similar for CT and RT, and smaller than for NT and NTSB, which were equal among them. This indicates that the physical condition of the soil under NT was different to the tilled treatments, regardless of residue management.

Finally, scores for PCA-F3 were only different for NTSB. Considering that N fertilization was the same for all treatments, differences in N properties among plots must be related to the organic fractions of N. This is supported by the observed differences in PON and enzyme hydrolysis of organic N (*i.e.*, urease activity). Alterations of the soil organic N components due to fire in Mediterranean pine forest A horizons have already been described (Knicker *et al.* 2003). It seems that residue burning under NT can also alter the type of N compounds in the upper soil layer. Burning crop residues under NT affects thus more the quality of organic matter and the N cycle than the physical quality of the soil. While the impact of burning biomass on the soil C cycle has been intensely studied, the impact of residues burning under NT on the N cycle remains mostly unknown.

Conclusions

The aim of this study was the evaluation of the effect of tillage and stubble burning under NT in the SQ of a Mediterranean semi-arid soil using PCA. We observed changes in the physical condition, organic matter and biological properties. As observed in other studies, the suppression of tillage (NT) resulted in more organic matter quality and different physical condition of the soil. Stubble burning under NT did not affect the physical quality of the soil, which was equal to NT, but induced changes in the organic matter. Organic matter in NTSB was similar to that under CT, and NTSB was different to all the other treatments in all the parameters studied in relation to the N cycle. If the best SQI are those more sensitive to changes in soil management, parameters related to organic matter quality, organic N and soil physical status should be evaluated to determine the effect of tillage practices and stubble burning under NT in this type of soils. In particular, the impact of stubble burning on the N cycle appears as a promising field of research.

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Soils as a target of anthropogeographic landscape changes in alpine areas (Dolomites, northern Italy)

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Abstract
The paper deals with the study of soils as a target of human-induced landscape changes in a typical alpine area, in the Dolomites region (Italy). A general soil survey allowed identification of different landform units in terraced soil systems. Forty representative soil profiles developed from limestone and under different vegetation cover (meadows, pasture, forest and agricultural land) were opened and sampled for routine analyses. Selected soil properties, both physical (soil structure, porosity, texture, root penetration depth, skeleton, water retention) and bio-chemical (pH, nutrient status, carbonate content, solute translocation, organic matter content and transformation, soil fauna activity) allowed assessment of the consistent effects of land use change on soils and soilscapes. Different evolutionary stages were recorded in the changing terraced landscape:

- little degradation under permanent meadow (older than 50 years);
- moderate degradation under forest (30 - 50 years);
- high degradation under abandoned forest (<50 years).

Dynamic soil properties, and the processes involved, proved greatly useful to understand transformation mechanisms and to assess possible consequences of land abandonment on soils and the whole environment.

Key Words
Terraced landforms, anthropogenic soils, land use, dynamic properties.

Introduction
The European Landscape Convention, subscribed by 27 States of the European Union (E. U. 2000), defines the landscape as a portion of land whose characteristics derive from natural and/or human factors, and which plays important general functions. The soil is defined, in turn, as part of the landscape, resulting from the synergism of different environmental factors, as expressed by the well known equation (Jenny 1941):
S= f(cl,o,r,p,t).
The increasing influence of human activities on the landscape and the transformation of the traditional socio-economy towards new development models has determined consistent effects on soils, particularly in mountainous areas. The progressive marginalisation of such areas as a consequence of land abandonment, in the whole alpine arc, has noticeable effects on soils and soilscapes, namely:

1. at ecological and environmental level:
    - Progressive re-naturalisation of open spaces;
    - Biodiversity reduction;
    - Slope instability;
    - Hydrogeological hazard;
    - Potential forest fire increase;

2. at economic level:
    - loss of economic capacity;
    - tourism negative impact;
    - loss of attraction capacity and appeal
    - difficult accessibility;

3. at social and cultural level:
    - know-how and *savoir-faire* losses;
    - cultural landscape disappearance;
    - natural resources banalisation;
    - wellness perception for both residents and tourists.

Consequently, the conservation of the soil resource is considered a fundamental item for human society, as it is stated in the Alps Convention (1991), "with the objective of protecting and restoring natural environment and landscape, in such a way to guarantee the ecosystem efficiency, flora and fauna conservation, unicity, diversity and beautifulness of nature and landscape". Indeed, soils present different aspects that qualify them as cultural heritage, assuming different values for each case:

- historical valency (paleosols and soils at archaeological site);
- scientific valency (soils that exemplify natural and/or anthropic processes);
- ecological valency (soils as parts of fragile ecosystems);
- aesthetic valency (soils that contribute to the amenities of the landscape);
- social-economic valency (touristic exploitation of marginal areas).

Therefore, soil study is a major concern in conservation of natural and cultural landscape, and is fundamental in land planning.
The objectives of this paper are:
- to identify different soil types of a changing alpine landscape;
- to highlight relationships among soil properties, land use changes and new landforms;
- to suggest new perspectives in the management of alpine terraced landscapes.

Materials and methods
The study area is located in the southern part of the Dolomites region, and covers a narrow belt about 50 Km in length; parent material is limestone; altitude varies between 400 and 1200 m a.s.l., with steep slopes, mostly terraced; the vegetation cover is a mosaic of meadows, pasture, mixed hardwood, or agricultural land. Presently, many terraced systems are abandoned and have lost their original function of protecting soil and land from degradation. Within the frame of the European project "Terraced Landscapes of the Alpine Arc" (ALPTER project 2006-2008), terraced landforms with different land use were identified by comparison of different sets of aerial photographs (1954-2006). Afterwards, 40 soil profiles were opened and sampled at different sites in artificially build terraces. Dynamic soil properties that may change in short time (Richter *et al.* 2007; Bellamy *et al.* 2005), both physical (e.g. soil structure, soil porosity, texture, root penetration capacity, etc.), and chemical (nutrient status, salinity, acidity, solute translocation, etc.) or biological (organic matter transformation, biological activity, etc.) were selected for each profile, following the DPSIR model, in order to understand how natural or man-induced environmental modifications (driving forces and pressures) may influence directly (impacts) the soil resource (state), in terms of soil responses to anthropic activities.

Results
Land use changes in the terraced landforms of Dolomites consist of a relevant decrease in total agricultural land (-40% in the last 50 years);
- permanent meadows and pasture present a significant decrease (3000 hectares, up to 15%) in the same period;
- forests have increased by 25% (28000 hectares) in the same period; however, most of the forested territory is not man-assisted during the expansion process, and this may determine significant relapses in ecosystem conservation;
- soil properties which change in a short time, as a consequence of changing land use, erosion processes, agricultural practices, etc. are: root penetration depth, skeleton percentage, water retention capacity, organic matter content and type, biological activity, fauna abundance and typology.

Three developing stages occur in terraced soils:
- little degraded soils under permanent meadow (>50 years old): deep umbric horizon, active pedofauna (mostly earthworms), strong crumb structure, good nutrient status. No structural landform degradation, no ecological degradation (INCEPTISOLS);
- moderately degraded soils under forest (<50 years old): reduced soil depth, shallow umbric horizon, slowed biological activity; reduced forest floor. No structural landform degradation, moderate ecological degradation, loss of biodiversity (INCEPTISOLS);
- strongly degraded soils under forest (<50 years old): shallow depth, abundant skeleton, little water retention, reduced biological activity, inconsistent forest floor, strong erosion phenomena. Structural and ecological degradation (ENTISOLS).

Conclusions

In the last two centuries, new agro-forestry systems, increased industrial activities, and the dichotomy between soil knowledge and land use, have determined intensive soil exploitation, determinating soil loss by erosion, chemical contamination, acidification, low fertility, or inhibiting its ecological functions (biomass production, biological filter, genetic reserve, habitat for flora and fauna). The reduced soil capability for agriculture and food production are evidence of the human contribution to the Global Soil Change, as suggested by Arnold et al. (1990), Richter (2007) and Zalaseiwich (2008). Although soils of artificial terraces occupy large areas all over the world, and have undergone to profound transformation with aggradation and degradation processes (e.g. cumulization, haploidization, pedoturbation, entisolization), only in recent years there has emerged the opportunity to revise their taxonomic allocation in a new category within the traditional soil classification systems (e.g. Soil Taxonomy, WRB), and one of the major types of ANTHROPOGENIC soils distributed on our planet (Dudal 2004; Icomanth 2007). Man becomes leading actor of land transformation, and plays an important part in the changing world, as well summarized by the Nobel Prize winner Paul Crutzen (2002) in defining the ANTHROPOCENE as the man-induced new geological era of this millennium. Differences among Anthropogenic and natural soils, therefore, should be evidenced, in view of a correct management of terraced areas. The study of the already recorded soil dynamic properties, and the processes involved, moreover, may be of great utility to understand possible consequences of land abandonment on soils and the whole environment. Although the effects of global soil change are presently little quantified, to understand how and at which intensity soils modified by man react, in tune, with the environment, is of capital importance to predict and quantify the anthropic effects in a short time and to realize effective management of rapidly changing ecosystems.

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The Brigalow catchment study: More than 20 years of monitoring water balance and soil fertility of brigalow lands after clearing for cropping or pasture

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Abstract
The Brigalow Catchment Study was established to determine the impact on hydrology and soil fertility when brigalow land is cleared for cropping or pasture. This paired catchment study commenced in 1965, when three catchments were selected in central Queensland, Australia, to represent the extensive brigalow bioregion of approximately 40 million hectares. Catchment hydrology was characterised during a 17-year calibration period (1965–81). In 1982, two of the three catchments were cleared, with one developed for cropping and the other sown to improved pasture. The third catchment was retained as an uncleared control. Soil sampling on 13 occasions from 1981-2007 allowed changes in soil fertility to be characterised. Land development for either cropping or grazing has doubled runoff and increased peak runoff rates. Deep drainage increased dramatically during land development and significant amounts of soil chloride were leached. This continues to occur under cropping. Cropping has resulted in a decline in soil organic carbon, total nitrogen and bicarbonate-extractable phosphorus. Grazing beef cattle on improved pasture however has maintained soil organic carbon and total nitrogen levels, but has shown a greater decline in bicarbonate-extractable phosphorus than cropping.

Key Words
Vertosols, Dermosols, Sodosols, buffel grass, modelling, virgin brigalow scrub.

Introduction
The brigalow bioregions of Queensland and New South Wales occupy 36.7 million hectares, stretching from Dubbo in the south to Townsville in the north. Since European settlement, 58% of this bioregion has been cleared. In 1962, the Brigalow Land Development Fitzroy Basin Scheme commenced, resulting in the clearing of 4.5 million hectares for cropping and grazing. This clearing represents 21% of all clearing in the brigalow bioregions and 32% of the Fitzroy Basin area. In order to quantify the effect of land clearing on hydrology and soil fertility, the Brigalow Catchment Study (BCS) commenced in 1965.

Methods
The BCS (24.81^o S, 149.80^o E) lies in the Dawson subcatchment of the Fitzroy basin, central Queensland, Australia. The region has a semi-arid, subtropical climate. Summers are wet with 70% of the annual average calendar rainfall of 720 mm falling between October and March, while winter rainfall is low. Rainfall is highly variable, ranging from 11 mm or less in any month, to 165 mm in one day. Annual potential evaporation is 2133 mm, and average evaporation is at least twice the average rainfall in all months.

The BCS is a paired catchment study consisting of three small catchments of areas 11.7-16.8 ha. There have been three experimental stages (Table 1). Mean slope of the catchments is 2.5%. Soil types in the catchments comprise associations of Black and Grey Vertosols, some with gilgais, Black and Grey Dermosols, and Black and Brown Sodosols (Isbell 1996). In their native state, the catchments were composed of three major vegetation communities, identified by their most common canopy species; brigalow (*Acacia harpophylla*), brigalow – belah (*Casuarina cristata*) and brigalow – Dawson Gum (*Eucalyptus cambageana*). Understoreys of all major communities are characterized by *Geijera* sp. either exclusively, or in association with *Eremophila* sp. or *Myoporum* sp. The catchments were good quality agricultural land, all equally suitable for cropping or grazing. The Study has been reported comprehensively (Cowie *et al.* 2007; Radford *et al.* 2007; Thornton *et al.* 2007; Silburn *et al.* 2009).

Table 1. The land use history of the three catchments of the Brigalow Catchment Study.

		Land use by experimental stage		
		Stage I	Stage II	Stage III
Catchment	Area (ha)	(Jan 1965-Mar1982)	(Mar 1982-Sep 1984)	(Sep 1984-Dec 2004)
1	16.8	Virgin brigalow scrub	Virgin brigalow scrub	Virgin brigalow scrub
2	11.7	Virgin brigalow scrub	Development	Cropping
3	12.7	Virgin brigalow scrub	Development	Improved pasture

The Stage I calibration phase (17 years) identified the inherent differences in catchment runoff characteristics allowing a calibration for empirical comparison between catchments. Three permanent soil monitoring sites (20 x 20 m) were established in each catchment: two on clay soil, in both an upper and lower-slope position, and the third on a Sodosol. Baseline measurements of soil fertility were taken in 1981.

Stage II commenced in March 1982, when C2 and C3 were developed by clearing vegetation with traditional bulldozer and chain methods. The fallen timber was burnt *in situ* in October 1982. In C2, residual unburnt timber was raked to the contour and burnt. Narrow-based contour banks were constructed at 1.5 m vertical spacing. A grassed waterway was established to carry runoff water from the contour channels to the catchment outlet. In C3, any unburnt timber was left in place, and in November 1982 the catchment was sown to improved pasture by distributing buffel grass seed (*Cenchrus ciliaris* cv. Biloela) on the soil surface. The second soil fertility assessment was undertaken in December 1982, soon after burning.

Stage III commenced in 1984. In C2, the first crop sown was sorghum (September 1984), followed by annual wheat for nine years. Fallows were initially managed using mechanical tillage (disc and chisel ploughs), which resulted in significant soil disturbance and low soil cover. In 1992 a minimum tillage philosophy was introduced and in 1995 opportunity cropping commenced with summer (sorghum) or winter (wheat) crops sown when soil water content was adequate. No nutrient inputs were used. In C3, the buffel grass pasture established well with >5 plants/m^2 and 96% groundcover achieved before cattle grazing commenced in December 1983. Stocking rate was 0.3-0.7 head/ha (each beast typically 0.8 adult equivalent), adjusted to maintain pasture dry matter levels >1000 kg/ha without feed or nutrient supplementation.

Soil fertility was assessed annually from 1981-1987 and then in 1990, 1994, 1997, 2000, 2003 and 2008.

Results

Runoff from the three catchments in their virgin state during Stage I averaged 34 mm/yr; approximately 5% of annual rainfall. Peak runoff rate averaged 3.4 mm/hr. Runoff data from Stage I was used to develop linear relationships to estimate runoff from C2 and C3 given known runoff from C1. This calibration was used to compare Stage III measured runoff from C2 and C3 with estimations of runoff had they not been cleared. This showed an increase of 42 mm/yr when brigalow scrub is developed for cropping and 38 mm/yr when developed for grazed pasture (Figure 1). Peak runoff rate from brigalow scrub increased to 6.9 mm/hr in Stage III however land development resulted in an additional 149% increase in peak runoff rate from the cropping catchment and 67% increase from the pasture catchment.

In 1981 prior to land development, soil chloride showed similar profiles across all sites, typically increasing to 0.4-0.6 m depth and then remaining relatively constant (Figure 2). Chloride mass in the clay soils was similar, with 25 t/ha of chloride to 1.5 m depth, however chloride mass in the sodosols was as low as 4.9 t/ha. During the land development phase (Stage II), the upper slope clay and sodosol sites in C2 showed significant loss of soil chloride, while all sites in C3 showed significant loss.

Subsequent reductions in soil chloride under cropping were only significant in the upper clay soil, while under pasture, no further significant change occurred (Figure 2). Chloride mass balance analysis indicates deep drainage of 0.17 mm/yr for clay soils and 0.26 mm/yr for Sodosols under virgin brigalow scrub. These drainage rates increased during the land development phase to 59 mm/yr for C2 and 32 mm/yr for C3. Since development, deep drainage has averaged 19.8 mm/yr under cropping and 0.16 mm/year under pasture.

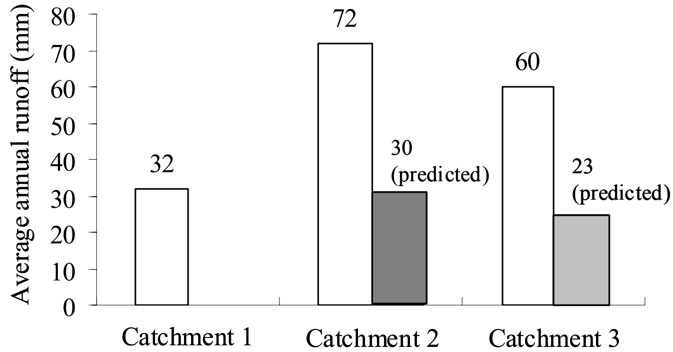

Figure 1. Observed runoff from the three catchments during the land use comparison phase (Stage III) of the trial (□), and the predicted runoff from catchments 2 (■) and 3 (■) had they remained uncleared. All data has been rounded to zero decimal places.

In their virgin state the three catchments had similar soil fertility. From 0-0.1m, levels of organic carbon (OC) (Walkley and Black) ranged from 1.8 to 2.2%, soil total nitrogen (TN) (Kjeldahl) from 0.18 to 0.21%, and extractable phosphorus (CP) (Colwell) from 10.3 to 11.0 mg/kg. No significant changes in OC levels occurred in the scrub or pasture catchments over the 26-year land use comparison; however, the cropping catchment showed a 48% decline in OC (Figure 3). Similarly, no significant changes in TN levels occurred in the scrub or pasture catchments, while the cropping catchment showed a 64% decline in TN. Burning of the pulled timber in C2 and C3 resulted in significant increases of CP, to levels of 36.8 and 34.3 mg/kg respectively. These levels decreased over the 26 years in both the cropping and pasture catchments by 45% and 65% respectively

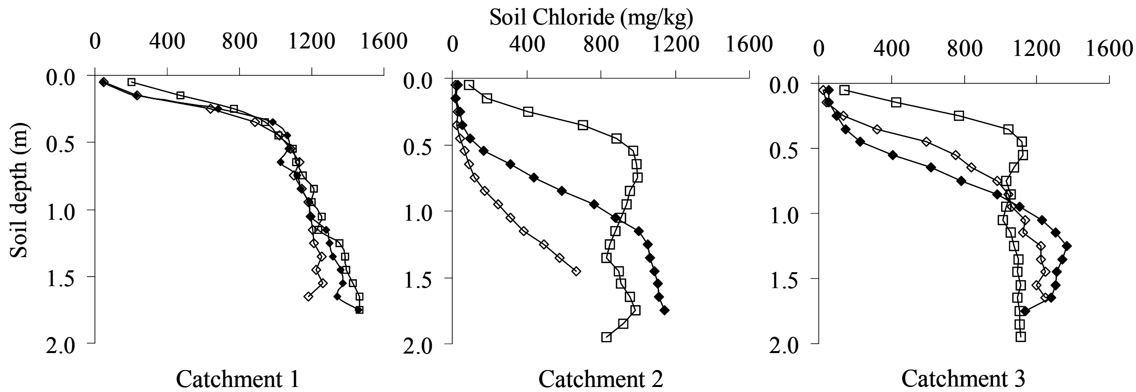

Figure 2. Average soil chloride profiles for the upper slope clay soil in each catchment pre-clearing (1981) (□), immediately after land development (1983) (♦) and after 16 years of land use (2000) (◊).

Discussion

At this site, land development for either cropping or grazing had significant effects on catchment water balance and fertility. A doubling of runoff and increased peak runoff rates under both agricultural systems resulted in increased risk of erosion and transport of nutrients and agricultural chemicals off-site. Agricultural production opportunities are also forgone as water for crop or pasture growth is lost.

An increased salinity risk is also of concern, primarily associated with the large increase in deep drainage and chloride leaching during the development phase of each land use and the ongoing deep drainage under cropping. The removal of chloride from the upper soil profile may however provide agricultural production benefits if initial chloride levels are a constraint to crop or pasture growth.

The three soil fertility parameters investigated all showed significant decline under cropping. Even if fertility decline is arrested via the application of fertiliser, the soil is unlikely to return to its virgin fertility level while continuing to be cropped. Modelling suggests that the application of nitrogen fertiliser to this system will improve TN levels, however limiting rainfall will not allow an increase in cropping frequency and hence dry matter production, so OC levels will not be improved (Huth *et al.* 2009).

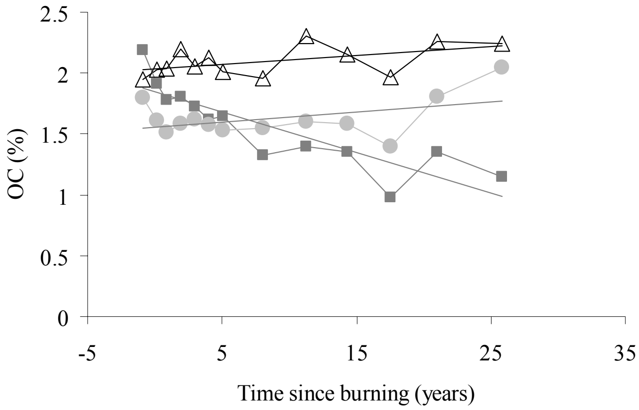

Figure 3. Average organic carbon levels (0-0.1 m) in the untreated brigalow scrub catchment (∆), linear trend (‒), over 26 years, compared to levels in the catchment developed for cropping (■), linear trend (‒), and the catchment developed for pasture (●), linear trend (‒).

Unlike cropping, grazed pasture appears capable of maintaining both OC and TN levels, however significant amounts of TN are likely to be held in a plant-unavailable form, which may be limiting to pasture growth. The greater decline in CP in the pasture catchment compared to the cropping catchment suggests that even though more P is removed in grain than in beef, continuous production of dry matter in the pasture system results in less available P than in a cropping system, where dry matter production occurs for only a few months of the year.

Conclusions

Development of brigalow lands for cropping and grazing has significantly altered water and nutrient balances, with increased runoff and peak runoff rates, increased drainage and decreased soil OC, TN and CP under cropping and decreased CP under pasture. Based on these indicators, pasture appears more analogous to the native brigalow landscape than cropping. The relevance of these findings to the larger brigalow bioregion will help to guide future investment in natural resource management. The length of record and breadth of data collected at this site can be considered a model in its own right, providing a point of truth for landscape and process modelling activities and a benchmark to assess the effects of slow, subtle and complex processes such as climate change on semi-arid subtropical Australian landscapes. To better facilitate these activities, open access to BCS data is available online at www.derm.qld.gov.au/science/projects/brigalow/index.html.

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The effect of climate and land use change on soil respiratory fluxes

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Abstract
The effect of stand age on soil respiration has been studied at three locations at the Dooary forest,, County Laois, Ireland. This forest is located on a wet mineral soil and the chronosequence includes a semi-natural grassland (T_0), a 6 year old Sitka spruce and a 20 year old Sitka spruce stand. Different rates of total soil carbon dioxide (CO_2) efflux have been found among sites where soil CO_2 efflux decreased with forest age. In this study afforestation has been shown to decrease soil CO_2 emissions. Soil respiration has been shown to be driven by certain climatic parameters such as air temperature, precipitation and soil water status. Changing future climatic conditions may therefore change the observed rates of soil CO_2 efflux. In order to investigate the impacts of changing soil water content on rates of soil respiration, precipitation exclusion shelters have been installed at the chronosequence sites. The results show, in the short term, that the impacts of a reduction in precipitation on soil respiration differ between the different chronosequence sites with a significant reduction observed in the six year old forest stands and no significant differences observed at the other sites.

Key Words
Soil CO_2 fluxes, climate change, land use change, forest and wetland.

Introduction
Information on carbon (C) sequestration and greenhouse gas (GHG) emissions associated with land use, land use change and forestry (LULUCF) is required for reporting commitments under the United Nations Framework Convention on Climate Change (UNFCCC) and the Kyoto protocol. Forest ecosystems are significant, long-term carbon sinks (Valentini *et al.* 2003), and under Article 3.3 of the Kyoto protocol, C sequestered through afforestation, which has occurred since 1990, may contribute towards a reduction in the national GHG emission inventory of a signatory country. Due to the rapid growth of the Irish economy over the past decade current GHG emissions are approximately 23.5% above the 1990 level (EPA 2007). During the same period Ireland's forestry sector has developed rapidly: afforestation in Ireland increased by ~186,000 ha between 1990 and 2000 (Joyce and O'Carroll 2002) and it is estimated that it will contribute to a reduction of 11 Mt carbon dioxide (CO_2) equivalents per year from the national GHG inventory (Black and Farrell 2006). Within terrestrial ecosystems the efflux from the soil is one of the largest carbon flux components, contributing between 20-40% of the total annual input of CO_2 into the atmosphere (Lankreijer *et al*. 2003). The accurate measurement of soil respiration is therefore necessary for a thorough understanding of forest carbon cycling and belowground metabolic activity.

Forest ecosystems can act as a sink or a source of atmospheric CO_2 on the basis of the net difference between the two fluxes of photosynthesis and respiration. The impact of afforestation of formerly arable land on ecosystem C dynamics needs to be better understood in order to maximize sequestration of atmospheric C (Saiz *et al.* 2006). Whilst there are some information on forest stand age effects on soil respiration (Ewel *et al.* 1986a; Irvine and Law 2002; Klopatek 2002; Saiz *et al.* 2006), there are significant gaps in scientific research on the impact of climate change. In particular the modification of water availability through changing seasonal drying and wetting cycles due to changes in climate and disturbances associated with forestry operations. Particular emphasis has recently been placed on the prediction of drier summers in Ireland resulting from climate change and the impact that this will have on vegetation and water resources (ICARUS 2008; c4i 2008). In addition little is known regarding the consequence of these changes for soil's sequestration potential. It is also important to consider the change in C source/sink capacity associated with afforestation as the vegetation changes from grassland to a forest canopy. This study investigates the impacts of land use change on soil respiration by measuring soil CO_2 emissions on a grassland and two different aged Sitka spruce (*Picea sitchensi*) forest stands. Moreover it will attempt to determine the effects of a projected reduction in the Irish summertime precipitation on soil respiration by using an ecosystem scale manipulation experiment that reduces the quantity of rainfall on experimental plots at the chronosequence sites.

Methods

Soil CO_2 flux measurements and the experimental design

A series of sites were selected to represent the typical land use change to forestry. This chronosequence included a semi-natural grassland (T_0), a 6-year-old and a 20-year-old Sitka spruce (*Picea sitchensi*).
The three sites were selected at Dooary, County Laois, and were close to one another to ensure the soil type (a surface water gley –representative of the typical post 1990 afforestation site) was as similar as possible. Regular measurements of soil respiration have been taken using an EGM PP Systems Infrared gas analyser with soil respiration chamber. Associated measurements including soil temperature and soil moisture content have been taken in conjunction with soil respiration measurements. The climate manipulation aspect of the experiment was achieved by installing precipitation exclusion zones at each chronosequences stage prior to the summer of 2009. The exclusion zones consisted of a shelter constructed using transparent plastic sheeting over a wooden frame, covering an area of 25 m^2. The polyethene sheeting was chosen so to reduce the absorption of radiation, ensuring that changes in respiration were due to water availability and not due to a reduction in root exudates. The roofs of the shelters were ~ 1.5 m above the ground and had sloping sides that did not reach the ground, thereby not increasing the ambient air temperature or obstructing air movements. There were four independent experimental plots at each site.

Results

Rates of soil CO_2 efflux were measured at each site from 30th April until 15th October 2009. The results show that the highest soil CO_2 emissions were measured in the grassland and were lower in the 6 followed by the 20 year-old forest stands. Emissions ranged between 0.4-1.7 g/m^2/h in the grassland; 0.4-1.2 g/m^2/h in the six years old forest and only 0.2-0.6 g/m^2/h in the twenty years old forest (Figure 1). The differences in fluxes between the grassland and the 20-year-old sites were found to be highly significant ($P < 0.001$).

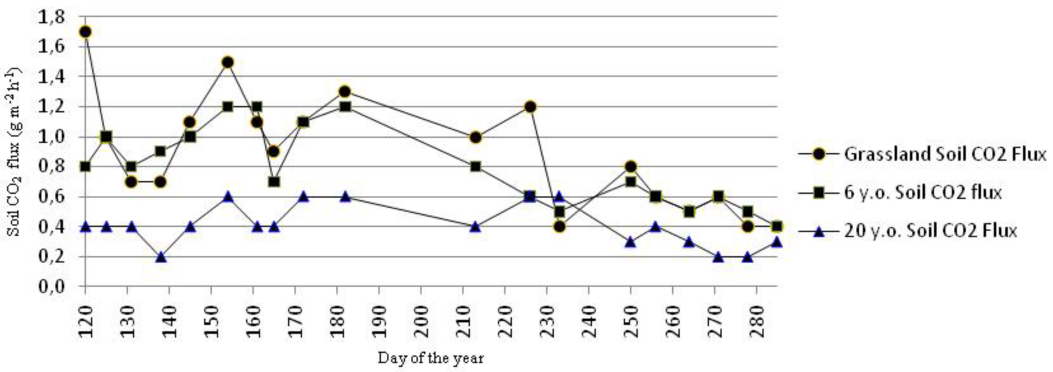

Figure 1. Mean soil CO_2 efflux at the Dooary forest chronosequence.

Soil temperature and soil moisture are driving forces in the production and emission of soil CO_2 (Raich & Schlesinger, 1992; Kirschbaum, 1995; Davidson *et al.* 1998). Measurements of soil temperature and soil moisture content were taken in conjunction with each soil CO_2 flux measurement and the relationship between soil CO_2 efflux and temperature and moisture content was analysed (Table 1).

Table 1. Relationship between soil CO_2 flux rates and soil moisture and temperature for the three chronosequence sites.

	Grassland	6 year old Sitka spruce	20 year old Sitka spruce
Soil Moisture	r^2= 0.1163 y = -0,0355x + 2,8963	r^2= 0.0033 y = 0,9169x + 53,744	r^2= 0.0208 y = 6,4785x + 38,834
Soil Temperature	r^2= 0.0475 y = 0,0328x + 0,4497	r^2= 0.0171 y = 1,2909x + 12,07	r^2= 0.4486 y = 10,418x + 8,169

As a result of the climate manipulation, soil CO_2 fluxes were generally higher inside the exclusion shelters at all sites (Figures 2-4). “In” and “Out” indicate the soil CO_2 efflux measurements taken inside and outside the rain out shelters respectively.

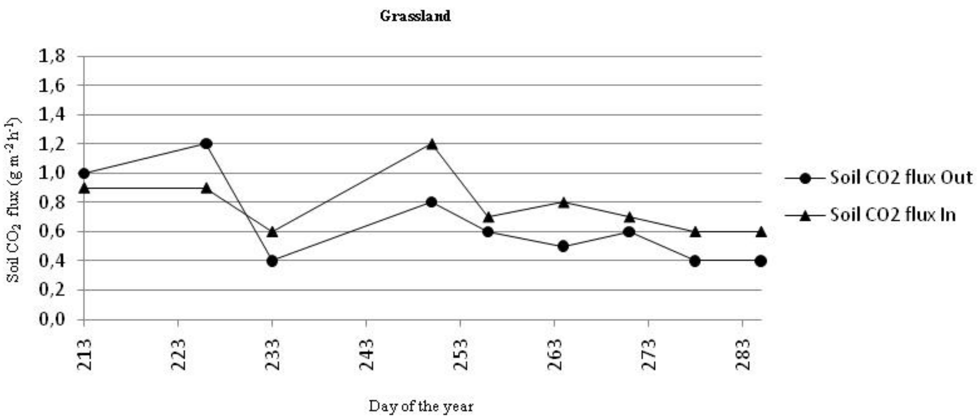

Figure 2. CO_2 fluxes (g/m^2/h) inside and outside the shelter on the grassland site.

The CO_2 emissions found inside and outside the shelters were not significantly different (P>0.05) and the soil water content measured inside and outside the shelters was not significantly different (P>0.05).
The grassland site is the most exposed of the chronosequence sites and the strong winds and the heavy rain may have compromised the efficiency of the rain-out shelters and the experiment. During the next months new shelters will be built and new measurements will be taken.

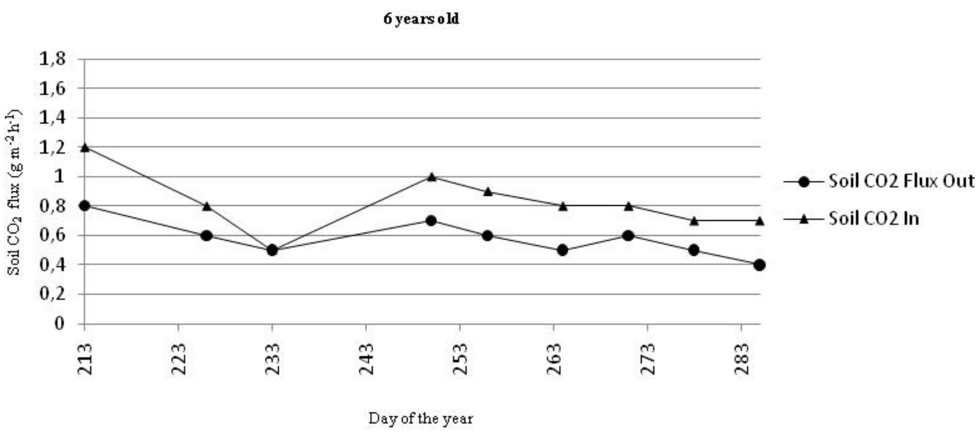

Figure 3. CO_2 fluxes (g/m^2/h) inside and outside the shelter on the 6 year old forest site.

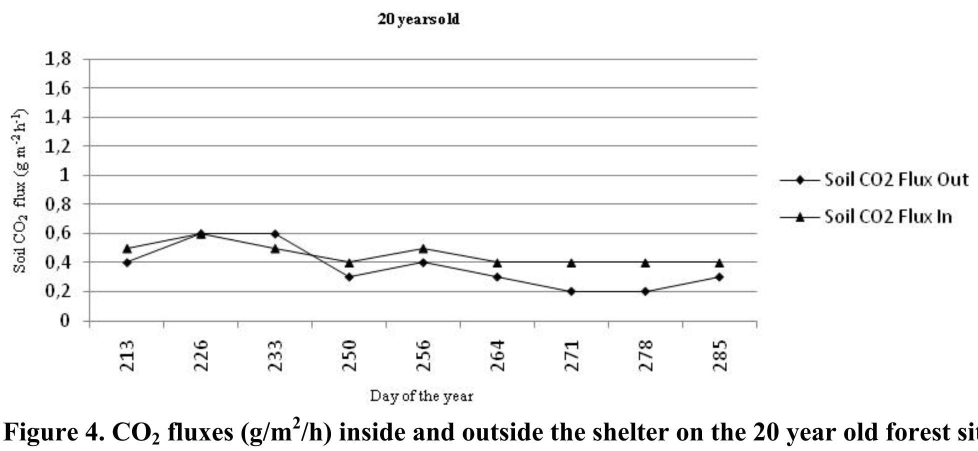

Figure 4. CO_2 fluxes (g/m^2/h) inside and outside the shelter on the 20 year old forest site.

The CO_2 emissions measured inside and outside the shelters were significantly different (P<0.01) in addition the soil water content measured inside and outside the shelters at the 6 year-old forest was highly significant (P<0.001). These results suggest that soil water content plays an important role on soil CO_2 production and emission. The CO_2 emissions found inside and outside the shelters were not significantly different (P>0.05) however the measured soil water content inside and outside of the shelters was significantly different (P<0.01). These results suggest that at this site the reduction in soil water content did not have a significant impact on rates of soil CO_2 efflux. The differences observed between the six and twenty year old Sitka spruce sites could be due, in part, to the differences in the composition of the understory vegetation at the two sites. Due to canopy closure at the twenty year old site understory vegetation is no longer present at the forest floor which may have reduced rates of soil CO_2 efflux due to changes in the autotrophic contribution to total soil CO_2 efflux. In order to test this assumption root exclusion cores have been installed inside and outside of the precipitation exclusion shelters at each site, which will allow total soil CO_2 efflux to be partitioned into its autotrophic and heterotrophic parts.

Conclusions

Afforestation results in a decrease in soil respiration although the reasons for this are not clear, as there is only a significant relationship with temperature in the oldest stand examined. Reductions in rainfall are predicted to have the greatest impact on older forest stands although perhaps surprisingly this may result in an increase in soil respiration. Overall these results indicate a complex relationship between soil respiration, afforestation and climate change that may depend on the climatic and site specific factors.

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The impacts of land use on the risk of soil erosion on agricultural land in Canada

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Abstract
Using established erosion models and national databases, A Soil Erosion Risk Indicator (SoilERI) was developed under the National Agri-Environmental Health Analysis and Reporting Program (NAHARP) in Canada to assess the risk of soil erosion in agricultural land from the combined effects of tillage, water and wind erosion processes. The indicator was build upon the Soil Landscape of Canada (SLC) polygon and was then aggregated to the province and national scale. It reflects the characteristics of the climate, soil and topography and responds to changes in land use over the 25-year period between 1981 and 2006. The results showed that the risk of soil erosion on Canadian cropland has steadily declined with time since the 1980s, largely due to the adoption of the conservation tillage, particularly no-till systems. However, there are still areas in every province with risks of unsustainable soil erosion. The risk of soil erosion was greatest under potato and sugar beet production and corn and soybean produced with conventional tillage. Serious erosion occurs on an important portion of cropland in southern Ontario and in Atlantic Canada. The information obtained in this study could help the decision makers to better target the hot spot of soil erosion in different scales and to design the best management practices for a given region.

Key Words
Soil erosion risk indicator (SoilERI), land use, agricultural land, Canada.

Introduction
Soil erosion is a major threat to agricultural sustainability in Canada. The loss of soil from current and past management is a major cause of low crop productivity and inefficient use of cropping inputs and can also have significant off-farm adverse impacts on the environment. Soil erosion occurs through three main processes: wind, water and tillage erosion. The combined effects of wind, water and tillage erosion pose a more serious threat than individual erosion processes. Management of the combined effects of wind, water and tillage erosion is required to maintain soil health. The risks of each component erosion processes and, therefore, the combined soil erosion are determined by the characteristics of the climate, soil and topography conditions and the land use. In a large scale, the characteristics of the climate, soil and topography conditions are relatively stable while the land use in agricultural land can change rapidly with time. This includes the change of crop types (e.g., from annual crop to forage) and tillage systems (e.g., from conventional tillage to no-till), both may cause the change of soil erosion risk with time. A Soil Erosion Risk Indicator (SoilERI) was developed under the NAHARP in Canada to assess the risk of soil erosion from the combined effects of all three forms of erosion processes. The SoilERI was designed to reflect the effects of land use change on the risk of soil erosion and is aimed at providing science-based agri-environmental information that can guide policy and program design (Lefebvre *et al.* 2005).

Methods
Soil erosion was calculated on the scale of Soil Landscapes of Canada (SLC) polygon (Figure 1). Each SLC polygon is characterized by one or more representative landforms, and each landform is characterized by hillslope segments (upper, mid and lower slopes and depression). Water, wind and tillage erosion rates were calculated for each segment. Other input data were obtained from established national databases or national surveys (Figure 1).

Individual erosion processes
Tillage erosion was calculated as the product tillage erosivity and landscape erodibility (Lobb *et al*. 2006):

$$A_{Ti} = ET \cdot EI \qquad (1)$$

where A_{Ti} is the rate of soil loss (Mg/ha/y), ET is tillage erosivity (Mg/m/%), and EI is landscape erodibility (% m/ha). ET was assigned to crops for given agricultural regions based on field experiments of tillage translocation studies carried out in various crop productions common in Canada. EL was calculated as a function of slope gradient and slope length.

Water erosion was determined using the Universal Soil Loss Equation (USLE):

$$A_{Wt} = R \bullet K \bullet LS \bullet C \bullet P \quad (2)$$

where A_{Wt} is the average annual water erosion rate, R is the climate factor, K is the soil erodibility factor, LS is the topography factor, C is the crop management factor and P is the supporting practice factor. However, adjustments have been applied to these factors based on intensive test runs of the Revised USLE version 2 program (RUSLE2) using data of the US counties along the US-Canada border (Li *et al.* 2008). The adjustment was made to capture the important advancements in water erosion science (e.g., the interactions between individual factors).

Wind erosion was calculated based on the Wind Erosion Equation (WEQ):

$$A_{Wd} = f(I,K,C,L,V) \quad (3)$$

where A_{Wd} is the average annual wind erosion rate, I is the soil erodibility index, K is the soil ridge roughness factor, C is the climate factor related to wind speed, air temperature and rainfall, L is the unsheltered distance across a field and V is the vegetative cover factor (Woodruff and Siddoway, 1965).

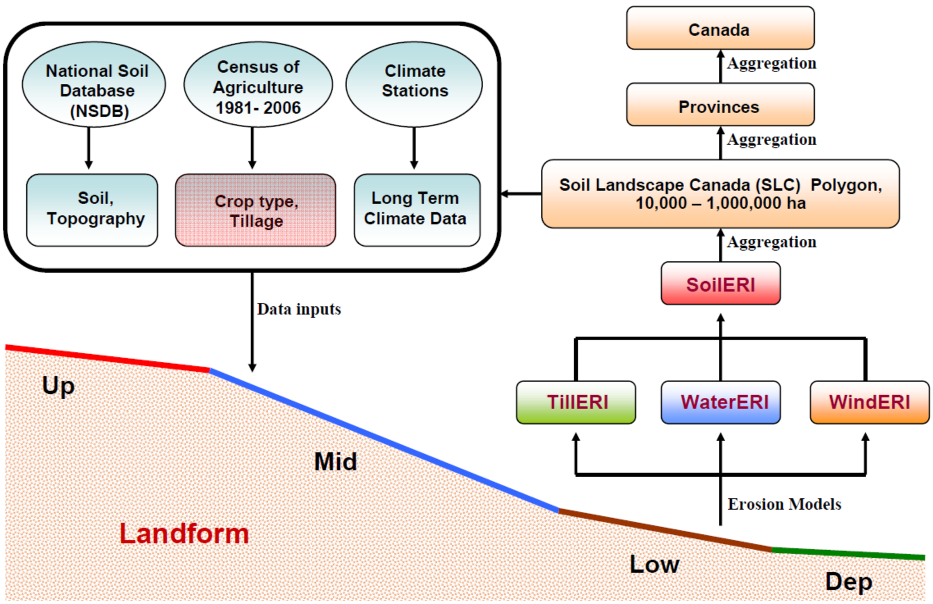

Figure 1. The framework of the Soil Erosion Risk Indicator (SoilERI).

The integrated Soil Erosion (SoilERI)

The integrated soil erosion was calculated as the sum of tillage, water and wind erosion for each landscape segments (Figure 1). The interactions (non-additive effect) between different erosion processes were not considered.

$$A_{Soil} = A_{Ti} + A_{Wt} + A_{Wd} \quad (4)$$

The soil erosion rates for individual segments were aggregated to the SLC polygon, province and national levels (Figure 1). The soil erosion rates were calculated for six years (1981, 1986, 1991, 1996, 2001 and 2006) corresponding to the Census of Agriculture in Canada and were grouped into six risk classes: negligible (< 3 Mg/ha/yr), very low (3 - 6 Mg/ha/yr), low (6 - 11 Mg/ha/yr), moderate (11 - 22 Mg/ha/yr), high (22 - 33 Mg/ha/yr) and very high (> 33 Mg/ha/yr). Areas in the very low risk class are considered capable of sustaining long-term crop production and maintaining agri-environmental health, under current conditions. The other four classes represent the risk of unsustainable conditions that call for soil conservation practices to support crop production over the long term and to reduce risk to water quality.

Results

The risk of soil erosion on Canadian cropland has steadily declined between 1981 and 2006 (Figure 2). The majority of this change occurred between 1991 and 2006. In 2006, 80% of cropland area was in the very low risk class (Figure 3). This is a considerable improvement over 1981 when only 47% was in this risk class. The cropland area in the higher risk classes each decreased by about one half during this time period, reaching a cumulative total of 20% in 2006. The integrated erosion risk indicator results paint a picture that is less positive than the results from the individual component indicators for water, wind and tillage erosion, but better reflects the actual risk of soil degradation by erosion. The improvement in soil erosion risk reflects reduction in all forms of soil erosion, however, the reduction in tillage erosion risk exceeded that of wind and water erosion.

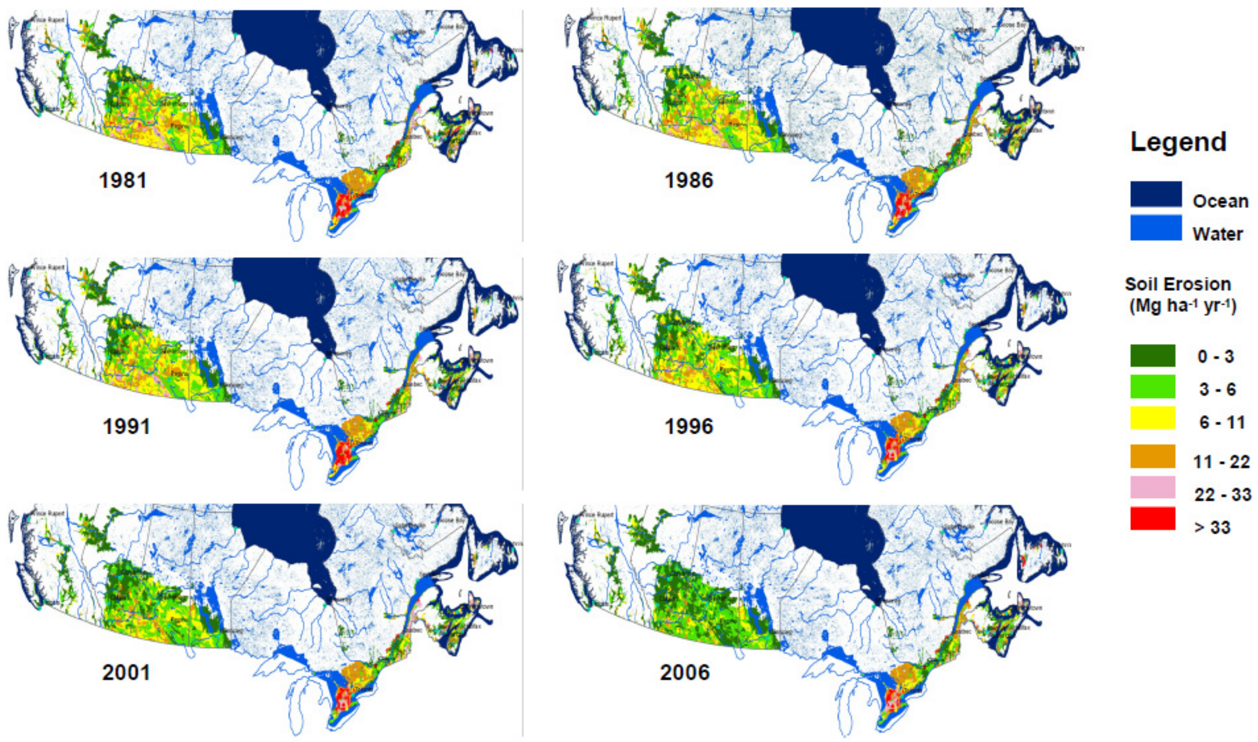

Figure 2. Soil erosion risk classes on agricultural landscapes in Canada from 1981 to 2006.

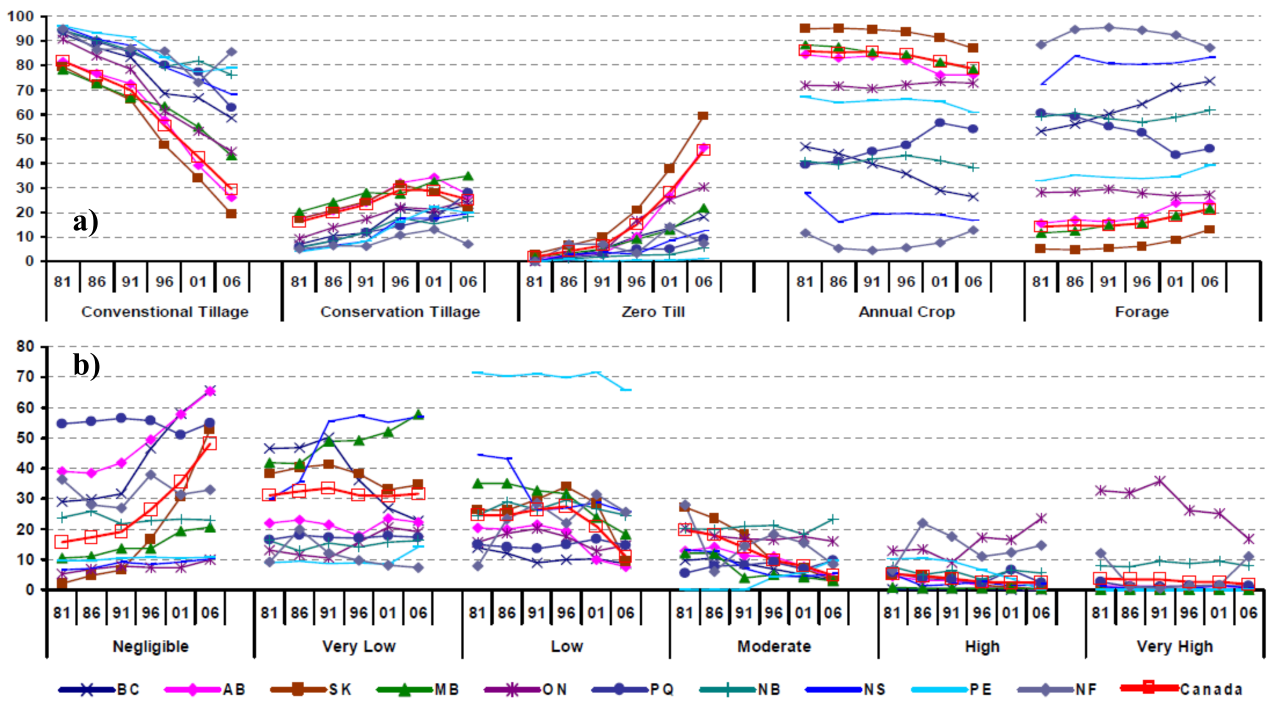

Figure 3. Shares (%) of cropland in Canada in: a) different tillage systems and major crop types; and b) different soil erosion risk classes

The decrease in all forms of erosion in Canada is largely due to the widespread adoption of conservation tillage, particularly no-till systems (Figure 3). Changes in share and mix of crops grown were less of a contributing factor. Crops requiring more intensive tillage, making them more erosive, such as corn, potatoes and beans, increased in area from 6% of cropland in 1981 to 13% in 2006. This uptrend was offset by the decrease in summer fallow, from 24% in 1981 to 9% in 2006, and by the increase in high residue crops requiring very little tillage such as alfalfa and hay, from 14% in 1981 to 21% in 2006. Although most crops

have seen a reduction in tillage intensity, the adoption of direct-seeding (no-till) in cereals has had the greatest influence on soil erosion, owing to the large share of cropland devoted to cereals.
Of the cropping systems across Canada, the risk of soil erosion was greatest under potato and sugar beet production where there is very intensive tillage and little opportunity to reduce the intensity through conservation tillage practices (data not shown). The cropping system with the next greatest risk of erosion is corn and soybean produced with conventional tillage, although there is a huge opportunity to reduce this erosion risk with conservation tillage. Of the soil landscapes across Canada, the risk of soil erosion is greatest on those with maximum slopes of 10% or more, especially those located in eastern Canada where climate produces a high inherent risk of water erosion. The most serious erosion concern occurs where the cropping systems with high erosion risks are practiced on soil landscapes with high erosion risks. This happens for an important portion of cropland in southern Ontario and in Atlantic Canada (Figure 2). However, there are areas in every province with risks of unsustainable soil erosion.
Detailed information such as the patterns and relative contributions of individual erosion processes could be used to delineate the hot spots of soil erosion in a given region, based on which to design the best management practices to reduce the risk of soil erosion.

Conclusions
Overall, the risk of soil erosion on Canadian cropland has steadily declined with time since the 1980s, largely due to the adoption of the conservation tillage, particularly no-till systems. However, there is still a large acreage of agricultural land in Canada with risks of unsustainable soil erosion. The detailed information could help the decision makers to better target the hot spot and, therefore, design the best management practices for a given region.

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Understanding the local pedological and ecological impacts of dust emitted from Cowal Gold Mine

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Abstract
The production of airborne particulates is a corollary of open-cut mining operations. Despite the best efforts of mine managers to minimise dispersal of these particulates, quantities of dust are inevitably carried beyond the mine margins and deposited in the surrounding landscape. Micromorphological and granulometric techniques were used to identify and subsequently isolate a likely aeolian component from the topsoils surrounding Cowal Gold Mine (CGM), in the Central West of New South Wales. Topsoil and subsoil samples were taken in four transects extending 12 km from the mine bund. An assessment of the geochemical properties of these samples will reveal the extent of contemporary aeolian deposition, and identify any trend consistent with CGM acting as a point source. In an effort to assess the likely ecological role of any mine dust accumulation, barley grass and rye grass were sampled along transects extending from the mine bund, and analysed for total metal concentration. Preliminary data infers that for a number of metals including lanthanum, molybdenum, lead and strontium, there is attenuation in plant uptake with increasing distance from CGM. A correlation of plant and soil data will extricate local soil factors and clarify the current role of mine-related dust, and the likely impacts of future accumulation of this dust.

Key Words
Aeolian dust, geochemistry, granulometry, pedogenesis, open-cut mining, micromorphology.

Introduction

The deposition of æolian dust is known to have been an active factor in the process of soil formation and landscape evolution on every continent. Given that the accession of dust to soils will impart properties not necessarily related to those developed by the weathering of underlying parent material, characterisation of the dust, its origin, nature and rate of deposition is essential to understanding soil development in receiving areas (Walker and Costin 1971). Dust particulates are categorised as being either naturally-occurring, agriculturally generated or formed by industrial activity Whilst the role of naturally-occurring dust particulates is relatively well understood in terms of pedogenesis and nutrient cycling in receiving areas, the function of anthropogenic dust is less well understood.

Dust produced by mining operations is likely to be enriched in metals not necessarily found in the overlying soil. Over time these dust-derived trace elements may accumulate in soils and sediments, and ultimately have a polluting effect. The greater environmental impact of any pollutant is realised once it becomes bio-available, and enters the food web (Agunbiade and Fawale 2009). Accordingly, this study will investigate potential metal accumulation by representative vegetative species in transects extending from the mine. This research will endeavour to give greater meaning to dust data in terms of the potential ecological effects on surrounding soil, water and biota, at the same time enhancing the understanding of any alteration to bulk soil attributes, pedological processes and properties.

Methods

Regional setting and sampling

Cowal Gold Mine (CGM) is located 50 km northwest of West Wyalong, in central western NSW. The mining bund protrudes into the high water mark of Lake Cowal, an ephemeral lake and wetland of national significance. Soil samples were taken on four perpendicular transects extending from the mining bund, as illustrated in Figure 1. These transects are 12.8 km in length, and are placed to encompass areas under the prevailing wind (south westerlies) across the mine as well as occasional wind receiving (and therefore potentially dust receiving) areas. It follows that this sampling system should allow for the characterisation of local soils in positions unlikely to be affected by mine dust and in positions likely to be affected by mine dust.

A parallel study by Hemi (2009) established a series of twelve Australian Standard dust traps in three transects extending from the CGM mine bund. Deposition data collected monthly over a two-year period indicates that despite considerable seasonal variation, the mine acts as a point source of dust; total deposition being greater to the east of CGM, and tapering with distance.

Figure 1. Soil sampling scheme for ecotoxicology study – 2009/2010. Sampling points are located where the concentric rings intersect the rays extending from the mining bund.

Sampling was carried out at intervals increasing exponentially with distance from the mining bund, such that a total of nine samples were taken along each of the four transects. At each point three samples were taken; a topsoil sample encompassing the top 10-20 mm of soil, a sample of the top 50 mm of the soil profile (representing the total expected depth incursion of recent dust accessions) and finally a subsoil sample at 400 mm depth. The deepest sample is assumed to be unaffected by contemporary (mine) dust and therefore provides data for comparison with overlying soil. The likely aeolian component, taken as those soil particles with diameters between 2 and 200 µm, was fractionated from the bulk soil material and used for subsequent analyses. A number of standard soil measurements were made on all bulk samples collected in the field including pH, EC and CEC.

Granulometry

The particle size distribution of bulk soil material from each sampling point was determined using the pipette method. The material 2 – 200 µm was analysed using a Coulter Multisizer 2, producing a high-resolution particle size distribution encompassing 256 size classes per sample (McTainsh *et al.* 1997). The high resolution analysis allows the identification of characteristic aeolian particle populations within the bulk soil, informing supposition about the distance travelled by the dust particles, and hence their likely source.

Mineralogy and elemental composition

Clay mineral suites of the segregated dust-sized fraction will be determined using X-ray diffraction (XRD) and then checked for dissimilarities observed between bulk soil and the dust sized fraction.

All soil samples have been digested using hydrofluoric acid, and await elemental analysis by ICP-MS. A comparison of the geochemistry of topsoil and subsoil samples will indicate whether a contemporary aeolian accession is implicit, and this data will then be rationalised with geochemical signatures of captured dust samples.

Morphology and micromorphology

Vertically-oriented thin sections (25 µm thick) were prepared using Kübiena tins of undisturbed topsoils collected at several points along each transect. Micromorphological features of these thin sections were examined under both plane-polarised and cross-polarised light using a petrographic microscope, with particular attention paid to the distribution, composition and morphology of any deposited dust.

Scanning Electron Microscopy (SEM) was also used to examine the morphology and composition of 'dust-sized' particulates occurring within the topsoil.

Collection and analysis of plant samples
Two annual grass species, Barley Grass (*Hordeum leporinum)* and Wimmera Rye (*Lolium rigidum*) were selected as 'indicator species' for the purpose of this study. In October 2008 and October 2009 ten individual plants of each species were harvested adjacent to existing dust traps that are established in transects extending from the mine bund. The plants were dried for 24 hours, ground and homogenised then digested using nitric/perchloric acid. The plant digests were analysed using inductively coupled plasma atomic emission spectroscopy (ICP-AES).

Results and Discussion
Preliminary analyses confirm the presence of a fine, seemingly allochthonous, and likely aeolian component in the topsoils surrounding CGM. The photomicrograph shown in Figure 2 was taken from a topsoil extracted approximately 1.6 km east of CGM, an area downwind of the mine. Figure 2 shows an explicit depositional layer (labelled as 'A'). The boundary between the existing solum (labelled as 'B') and the deposited material ('A') is clearly defined in terms of elementary fabric, skeleton grains and the colour of the fine matrix material. Given that this sample is effectively taken in a lake-bed, the depositional layer could theoretically be lacustrine clay, although reason would suggest otherwise. Lake Cowal has not held water in a number of years, and in that time the soil surface has experienced extreme desiccation, a resultant depletion of vegetation and consequently some significant disturbance. Furthermore the heavy, self-mulching vertosol soils which cover the lake-bed would have since incorporated any thin lacustrine deposit laid down during the last flood. Therefore the depositional layer is assumed to be aeolian. The source of this aeolian material is a matter to be determined, although the sizes of detrital grains in the depositional layer (20-150 µm) are consistent with that which may plausibly have travelled from the mine.

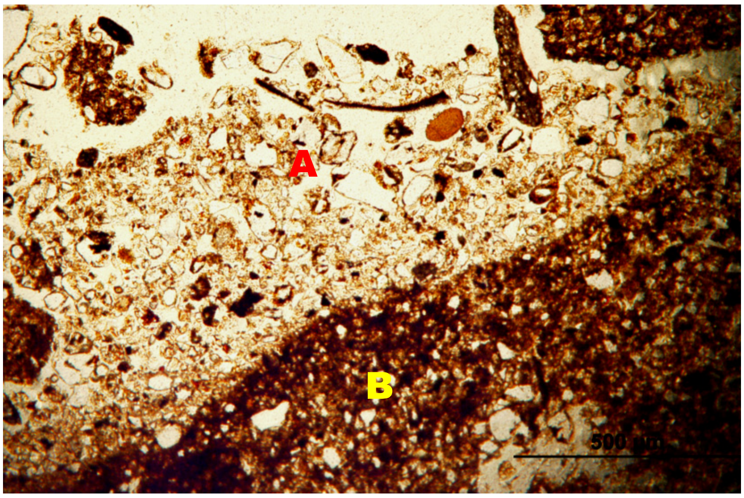

Figure 2. Photomicrograph of topsoil 1.6 km east of the mine bund, viewed under plane polarised light.

Preliminary results from the ICP-AES analysis of barley grass samples collected in October 2008 are shown in Table 1. The most consistent feature appears to be that background samples generally report much smaller values than those plants sampled downwind of CGM. Whilst this anomaly may be a related to an absence of dust accretion, it might also be accounted for by differing soil types. An examination of total metal concentrations in the topsoil and subsoil at plant sampling sites will be necessary before this data can be decisively interpreted.

This preliminary data does infer that for some metals at least, there is attenuation in plant uptake with distance from the mine. This trend certainly holds for lanthanum, molybdenum, lead and strontium concentrations in both rye and barley grass and the trend is consistent with an aeolian point source for these metals. Geochemical data from dust samples collected along equivalent transects, as reported by Hemi (2009), sustains the notion that these metals are supplied by CGM. Hemi (2009) describes a pattern of decreasing concentrations of a number of metals, including the lanthanides cerium and lanthanum as well as molybdenum, strontium and vanadium, with increasing distance from CGM, emulating the observed pattern of metal uptake by barley and rye grass.

Table 1. ICP-AES determined concentrations of selected metals in Barley Grass (*Hordeum leporinum*), relative to distance from the mine pit centre. Background samples taken 10.4 km south west of CGM.

Distance from centre of pit (km)	Metal Concentrations (mg/kg)										
	Ce	**Cr**	**Cu**	**La**	**Mo**	**Ni**	**Pb**	**Sr**	**Ti**	**V**	**Zn**
0.7	1.44	30.32	4.43	1.19	1.57	12.91	1.09	20.64	5.68	1.57	18.88
1.6	1.51	19.77	5.98	1.21	0.89	8.95	0.94	15.64	4.00	1.32	18.69
3.9	0.81	20.34	5.73	0.91	0.79	9.63	1.01	13.82	1.43	1.06	28.90
7.8	1.31	12.28	5.96	1.05	0.87	6.12	0.69	10.47	4.59	1.24	15.33
Background	0.86	7.75	3.41	0.69	0.14	4.92	0.46	16.17	5.79	0.40	27.74

Conclusions

Preliminary analyses have confirmed the accumulation of an aeolian component in the topsoil surrounding Cowal Gold Mine. Subsequent analyses of geochemistry and granulometry will allow apportionment of this aeolian material to a mine or non-mine source.

Plant uptake of a number of heavy metals appears to decline with increasing distance from the mining bund, suggesting that CGM may be acting as a point source for dust-derived trace elements. Pending investigation of local soil characteristics will allow the role of point source dust to be better clarified.

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Use of ecological agriculture as soil management system to improve soil properties and to mitigate greenhouse effect

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Abstract
Over recent decades the quality of agricultural soils has suffered deterioration due to inappropriate agricultural management. This management has caused, among other negative effects, the increase in salinity, erosion and contamination, and the liberation to the atmosphere of great amounts of organic carbon due to a higher mineralization. As a result, there has been growing interest in so-called sustainable agriculture, which takes into account not only productivity, but also the quality of the soil and environmental protection. For this study two fertilisation trials were carried out. The first one was carried out according to the stipulations of European regulations on practices of organic agriculture, while the second one was conducted in accordance with conventional agricultural practices in the area. Over a three-year period, samples of the arable soil layer were taken in order to analyse the soil properties. The plot under organic agriculture techniques management showed better physical conditions for crop development, greater quantities of organic matter and nitrogen, greater cation exchange capacity and greater phosphorous availability than the conventionally managed plot. These results suggest that, although the trial period was only three years, the agronomic model based on organic agriculture has a permanent beneficial effect on soil properties and contributes soil function as C sink.

Key Words
Organic agriculture, fertilisation practices, soil quality, C sink.

Introduction
Soils represent the largest terrestrial stock of C, holding approximately 1,500 Pg (10^{15} g) C in the top metre and 2456 Pg at a depth of 2 m (FAO 2002) This carbon storage in soils makes of them one of the most important CO_2 deposits in our planet. Thus, agricultural practices which minimize organic matter mineralization or even increase OC contents will not only be relevant regarding the influence on soil properties, but also by the positive role in mitigating greenhouse effect .

Since the 1990's numerous studies have shown that agricultural activity whose only aim is to obtain higher profits has led to the progressive degradation of the natural environment and the subsequent loss of productive capacity. Soil degradation due to salinity is especially outstanding, as it affects 30 % of all agricultural use soils.

Mediterranean soils usually have low levels of organic matter. This reduction in organic matter leads to a deterioration of the soil physicochemical properties, and consequently to the loss of long-term productivity. Agronomic practices, such as type of crop, crop rotation and management of residues have a considerable bearing on the content of organic matter in the soil and especially on the fractions which are most sensitive to practice-induced changes. The growing interest in rational soil use implies the preservation of organic matter and its associated microflora. In this sense, plots under ecological farming methods management as opposed to conventional ones have shown to obtain higher levels of organic carbon (OC), total nitrogen (TN) and cation exchange capacity (CEC).

The present work has studied the evolution of several soil chemical parameters depending on the method of crop management employed over a period of three years. It aims to show the differences observed in soil management and to determine the measured parameters usefulness as indicators of soil quality as well as the soil carbon capacity given the pressing need to establish strategies of soil preservation and sustainable use. The properties/parameters selected for evaluation are those which may change according to the use and management over relatively short periods of time, and as such are suitable indicators of soil quality.

Methods

Location and experimental design

The experiment was carried out on a trial plot in southeast Spain (Figure 1), dedicated to green crops where a high-frequency local irrigation system was used In each subplot three blocks of replication were stablished, following a "complete random blocks" design.

Figure 1. Experimental location.

Three fertilization trials and three irrigation trials were carried out simultaneously. Regarding the fertilization trials, the first one (F1) consisted of an organic amendment (sheep manure) with a 1,7kg $m^{-2}\cdot$ $year^{-1}$ ratio; the second fertilization essay, used as control was made according to a conventional cropping system in the area, while the third fertilisation trial (F3) was a commercial organic amendment, suitable for ecological agriculture and made up with sheep manure and vegetable waste compost, which was applied, with a 0,7 $kg/m^2 year^1$ ratio. Attending to the irrigation trials, three different doses were used: a deficitary dose (A), an optimal dose (B) and an excedentary dose(C). The treatment matrix is shown in Figure 2.

	F1	F2	F3	Replications	Total blocks
A	AF1	AF2	AF3	3	9
B	**BF1**	**BF2**	BF3	3	9
C	CF1	CF2	CF3	3	9
Replications	3	3	3	9	
Total blocks	9	9	9		**27**

Figure 2. Experimental design.

Arable layer samples (0-25 cm) were taken monthly during celery (APIO? – no será CEREAL) crop, being each sample a compound obtained in four different points of each plot. The samples were air dried and sifted to 2 mm for subsequent analysis in the laboratory. This paper describes only the evolution of soil properties in two treatments (BF1 and BF2), since these were the most different crop management system. Consequently, any existing difference in properties and evolution of soil between both fertilization essays would appear with more intensity.

General soil properties

The experiment was carried out on a soil with the profile *Ap1-Ap2-Bw-Ckm*, which meets the requirements to be classified as Palexeroll Petrocalcic (SSS 2006), Kastanozem Petrocalcic (FAO-ISRIC-IUSS, 2006). Soil had a loamy texture and OC content of A horizon was moderate. pH was basic (8,2), and no salinity problems were observed (EC in saturation extract lower tan 2 dS m^{-1}) it also had high concentrations of calcium carbonate (> 400 g/kg).

Chemical analyses

Organic Carbon (OC) was determined following the method of Anne modified by Duchafour (1970). Organic matter (OM) was calculated indirectly from the OC content, multiplying it by 1.72, the value expressed as OC in $g\cdot kg^{-1}$. Total nitrogen (TN) was determined by using the Kjeldahl's method as described by Duchafour (1970). Assimilable phosphorous (P) was determined by using the method of Watanabe and Olsen (1965) extracting P with 0.5M solution of $NaCO_3H$ and photocolorimetric determination of the amonic phosphomolybdate in a Philips PU 8625 UV/VIS spectrophotometer. Cation exchange capacity (CEC) and the Na, K and Mg exchangeable bases were determined by using Chapman's method (1965). Na, K and Mg were quantified by atomic absorption spectrometry, while the CEC was obtained by determining the ammoniac N with H_2SO_4 0.02 N using Bromocresol Green-Methyl Red as indicator.

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Statistical methods.
The data were statistically treated using the R software. When conditions of normality and homoscedasticity could be assumed, an ANOVA test was carried out to compare means; otherwise a non-parametric test of comparison of ranges was applied (Wilcoxon Test)

Results
As shown in Table 1 and Figure 3 and 4, content in Organic Carbon (OC), Total Nitrogen (TN), C/N ratio and Olsen Phosphorous (P) were significantly higher in treatment F1 than in the control plot F2. The increase of OC and P in F1 are related to the addition of organic matter, which stimulates microbian decomposition, and the positive action that humic compounds have on the availability of P, due to the contribution of organic phosphorous which when degraded by micro organisms in the soil, releases phosphate compounds (Tarafdar and Claasen 2005). Capriel (1991) and Mäder *et al*. (2002) reached the conclusion that CO content decreased less over time in plots under organic treatment than in conventional ones. Regarding the influence that ecological agriculture could have on soil as carbon sink, and in light of the results, use of manure amendments can be suggested to offset carbon emissions, especially in arable land, according to Article 3.3 and 3.4 of the Kyoto Protocol (Ogle *et al.* 2003; Smith and Powlson 2000). In this sense, these management practices imply an increase of 32% SOC in comparison with conventional agriculture (88,7 10^6 g CO_2/ha).

Table 1. Evolution of soil constituents and properties over 2005-2007.

Variables		2005	2006	2007	Mean	SD	N	P value
OC	F1	20,7	22,3	26,7	23,2	4,4	33	0,000***
	F2	17,8	17,8	16,8	17,5	1,6	33	
TN	F1	3,0	2,6	3,2	2,9	0,4	33	0.0116*
	F2	2,9	2,1	2,3	2,4	0,5	33	
C/N	F1	6,9	8,6	8,3	8,0	0,7	33	0.032*
	F2	6,1	8,5	7,3	7,3	0,8	33	
P	F1	44,3	50,8	41,0	45,4	4,1	33	0.00***
	F2	27,1	31.2	23,1	27,1	3,3	33	
CEC	F1	14,1	14,6	14,5	13,3	1,51	33	0,022*
	F2	11,8	13,3	12,2	12,5	1,6	33	
Na	F1	0,54	0,48	0,26	0,43	0,16	33	0,967
	F2	0,45	0,36	0,24	0,35	0,13	33	
K	F1	0,77	0,93	1,09	0,93	0,16	33	$1{,}11\cdot10^{-5}$***
	F2	0,56	0,45	0,29	0,44	0,14	33	
Mg	F1	0,96	0,88	0,82	0,89	0,07	33	$1{,}9\cdot10^{-7}$***
	F2	0,86	0,85	0,64	0,78	0,12	33	

CO, TN, Na, K, Mg (g/kg); P (mg/kg); CEC ($cmol_{(+)}$/kg)

The difference in N contents can be related to the decrease in F2, possibly due to leaching. It can be said, therefore, that the addition of sheep manure as organic amendment diminishes the loss of N (Hossain *et al.* 2003), but if N is applied in mineral form it is more likely to be lost to a greater of lesser extent depending on the soil characteristics.

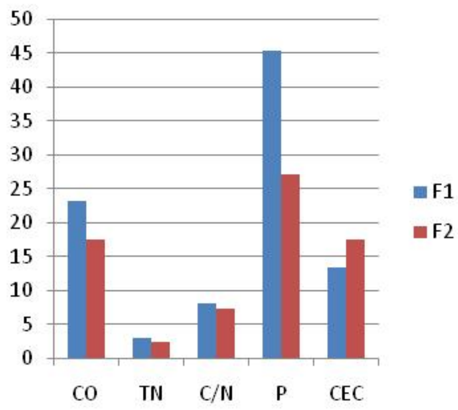

Figure 3. Average values of soil properties and constituents.

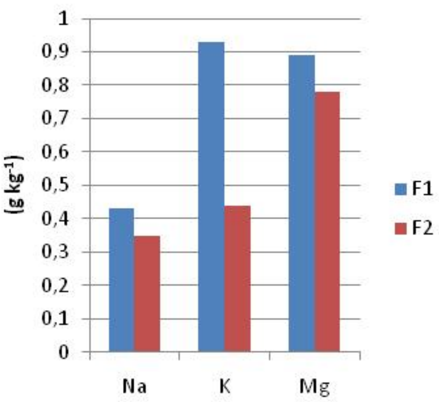

Figure 4. Exchangeable bases.

The low C/N ratio (less than 10 in all samples), lower than expected for calcareous mull humus, indicates that mineralisation of the organic matter prevails over humification, which gradually exhausts the soil N content unless it is periodically restored by applying additional sources. Cation exchange capacity (CEC) showed significant differences between treatments ($p<0.05$). These results confirm those obtained in other studies (Bending *et al*. 2004; Liu *et al*. 2006) which describe an improvement in the soil's CEC due to the addition of organic matter when compared to soils treated with conventional fertigation treatments. Contents of exchangeable sodium, magnesium and potassium are greater in the organically managed plot F1 than in the conventionally managed one F2, although differences found in Na contents between both treatments had no signification. These results confirm those obtained by Bulluck *et al*. (2002), and Morari *et al*. (2008) related with the improvements in the levels of nutrients linked to the addition of organic matter.

Conclusions

Considering the results, ecological agriculture is a soil and crop management system which increases the soil content in organic matter as well as certain nutrients such as nitrogen, phosphorous, magnesium and potassium compared to the plot treated with inorganic fertilisers. Changes were also observed in the soil properties, with an increase in the cation exchange capacity and the C/N ratio in the plot under manure treatment. Likewise, the results suggest that ecological agriculture practices increase soil carbon sequestration capacity.

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