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Afforestation of agricultural land with spotted gum (*Corymbia citriodora*) increases soil carbon and nitrogen in a Ferrosol

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Abstract
Planting forests onto marginal ex-agricultural land may provide a relatively cost-effective way of creating a carbon sink where more CO_2 is being removed from the atmosphere than is being released (sequestration), while simultaneously improving soil organic carbon (C) and nitrogen (N) stocks (organic matter) and fertility. Soil C and N stocks were measured in spotted gum (*Corymbia citriodora* spp. *variegata*) plantations (C_3 vegetation) established on ex-pasture (C_4 vegetation) sites compared with those in adjacent native vine scrub, pasture and peanut cropping in southeast Queensland (26°39'S, 151°45'E), Australia. The contribution of spotted gum to soil C was assessed using the natural ^{13}C isotope dilution technique ($\delta^{13}C$). Soil C and N concentrations were greater under the 4-year-old spotted gum plantation than either the adjacent grazed pasture or peanut cropping soil and similar to the original native vine scrub in the 0-0.3 m depths. Similar trends were observed in the total soil N. The quantity of C sequestered belowground was estimated to be approximately 240 Mg CO_2-e/ha in the entire 1.1 m soil profile after 25 years, indicating that afforestation may provide carbon offsets as an important source of future income as well as improved soil fertility (increased soil N) in the degraded agricultural lands of the South Burnett region in Queensland, Australia.

Key Words
Stable isotopes, $\delta^{13}C$, soil C sequestration, afforestation, hardwood plantations.

Introduction
Land-clearing for agriculture has invariably led to the loss of soil carbon (C) stocks in southern Queensland, as well as a decline in soil fertility and quality (Dalal and Mayer 1986; Bell *et al.* 1995; Saffigna *et al.* 2004). Due to the declining soil fertility and the introduction of the Vegetation Management Act (1999) that prevents tree clearing in Queensland, Australia, hardwood plantations are currently being established on degraded ex-agricultural soils. Hardwood plantations offer a viable land use for soils of the South Burnett region, Queensland, that have been cleared of native vegetation, but are marginal for agricultural production because of low fertility status or slope. Afforestation of degraded ex-farmlands with hardwoods can increase soil C (soil C sequestration) in subtropical regions (Paul *et al.* 2002; Maraseni *et al.* 2008), as well as improve soil fertility (increase soil N) and soil quality.

Currently the range of soil data to support changes in soil C and nitrogen (N) stocks estimations is limited. While it can be inferred that the impact of hardwood plantations on soil quality and C sequestration will be positive (Paul *et al.* 2002; Lima *et al.* 2006), the magnitude of this impact has not been quantified in southeast Queensland. One of the most common limitations to obtaining good data is that trees are planted before a comprehensive evaluation is made of the soil C and/or N stocks in the various land systems to be planted. In addition, there is generally no native vegetation area close by to provide the prevailing ecosystem-induced soil C and N levels. The establishment of more than 9000 ha of hardwood plantations in the South Burnett region, led us to investigate similar native vine scrub, pasture and peanut cropping areas on Red Ferrosols near Taabinga Village (26°34'59"S, 151°49'59"E), with the added land-use of an adjacent 4-year-old hardwood plantation that was planted onto ex-pasture.

The objectives of the study were to: (a) assess soil C and N changes down to 1.1 m depth for the four different land uses; (b) determine the light fraction C and N to assess the changes in soil C and organic matter quality; and (c) derive the turnover rate and time of hardwood-derived C and N in soil under plantation using delta (δ) ^{13}C isotopic natural abundance of soil organic matter to differentiate from C_4 pasture derived C.

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Methods

Land Uses

The four sites used in this study were located at Taabinga near Kingaroy (26°35'S, 151°50'E) in the inland South Burnett region of southeast Queensland, Australia. The soil is classified as a Red Ferrosol according to the Australian Soil Classification of Isbell (2002) or a Tropeptic Eutrustox (i.e. Oxisol) by the Soil Survey Staff (2006). Adjacent areas of Red Ferrosols under the following four land uses were sampled for this study:

- Site 1 - undisturbed native vine scrub (26°39'56"S, 151°45'08"E);
- Site 2 - wiregrass (*Aristida ramose*)/ Rhodes grass (*Chloris gayana*) pasture (26°39'38"S, 151°45'18"E);
- Site 3 - peanut (*Arachis hypogaea*)-maize (*Zea mays*) cropping (26°39'42"S, 151°45'28"E); and
- Site 4 - 4-year-old spotted gum (*Corymbia citriodora* spp. *variegata*) plantation (established November 2001) on ex-pasture (26°39'53"S, 151°45'09"E).

Soil and Plant Sampling

Soil samples were collected in April 2005 using a 44 mm diameter soil coring tube driven by a hydraulically operated soil sampling rig. However, in the native scrub a hand auger (42 mm diameter) was used because the vegetation was too dense to access with the soil sampling rig. The National Carbon Accounting System (NCAS) methodology was used for soil sampling (McKenzie *et al.* 2000). Each plot was divided into four quadrats and within each quadrat a sampling point was randomly located. At each sampling point, one core was taken to 1.1 m soil depth and then four adjacent cores were taken to 0.3 m depth. Each main core was divided into 0-0.05, 0.05-0.1, 0.1-0.2, 0.2-0.3, 0.3-0.5, 0.5-0.7, 0.7-0.9 and 0.9-1.1 m depths in the field, transferred to a plastic bag and sealed. The smaller cores were divided into 0-0.05, 0.05-0.1, 0.1-0.2, and 0.2-0.3 m depths and bulked with the main core samples at the corresponding depths. All soil samples were then transported to the laboratory for further analysis.

Analysis

Soil materials were air-dried and then sieved using a 2-mm sieve. Coarse material (stones and roots) was separated and masses were recorded. Representative sub-samples of the soil and light fraction (LF) of soil were fine-ground (<0.5 mm) for soil C and N analyses. Light fraction C (LFC) and N (LFN) was determined using sodium polytungstate solution (1.6 Mg/m^3 density) as described by Dalal *et al.* (2005a,b). Total soil organic C (TOC), soil total N, LFC, LFN, and natural abundance ^{13}C of soil, LF and litter samples were determined using an Isoprime isotope ratio mass spectrometer (IRMS) coupled to a Eurovector elemental analyser (Isoprime-EuroEA 3000) with 10% replication. The isotope ratios were expressed using the 'delta' notation (δ), with units of parts per thousand (‰), relative to the marine limestone fossil Pee Dee Belemnite standard (Craig 1953) for $\delta^{13}C$ using the relationship in equation 1 below:

$$\delta^{13}C\ (‰) = (R_{sample} / R_{standard} - 1) \times 1000 \qquad (1)$$

where R is the molar ratio of $^{13}C/^{12}C$ of the sample or standard (Ehleringer *et al.* 2000).

Results

Table 1. Key properties of the top layer of the Red Ferrosol under different land uses (n=4).

Property	Crop (0-0.05 m)	Pasture (0-0.1 m)	4yo plantation (0-0.1 m)	Native vine scrub (0-0.05 m)
pH (1:5 water)	4.56	5.10	4.56	4.46
EC ($\mu S/cm^1$)	92.5	41.0	57.7	169.6
C/N ratio	11.5	11.8	12.8	13.8
Sand (%)	58.0	46.0	45.0	49.4
Clay (%)	34.0	37.3	32.2	29.8
Silt (%)	8.0	8.0	10.0	19.2
Bulk Density (g/cm^3)	1.62	1.21	0.95	1.16

The distribution of TOC and soil N concentrations decrease with depth and vary with land use but below 0.5 m depth the TOC and soil N concentrations were similar across land uses except in the cropped soil (Figure 1). In the top soil layer (0-0.05 m depth) at Taabinga, the native vine scrub exhibited the greatest soil TOC concentration with 4.9%; followed by the 4-year-old plantation at 3.9%, pasture at 2.8% and lastly, the cropped site with 1.0% TOC. Soil C and N stocks were also calculated on equivalent soil mass basis, taking into account the changes in bulk density following land use change.

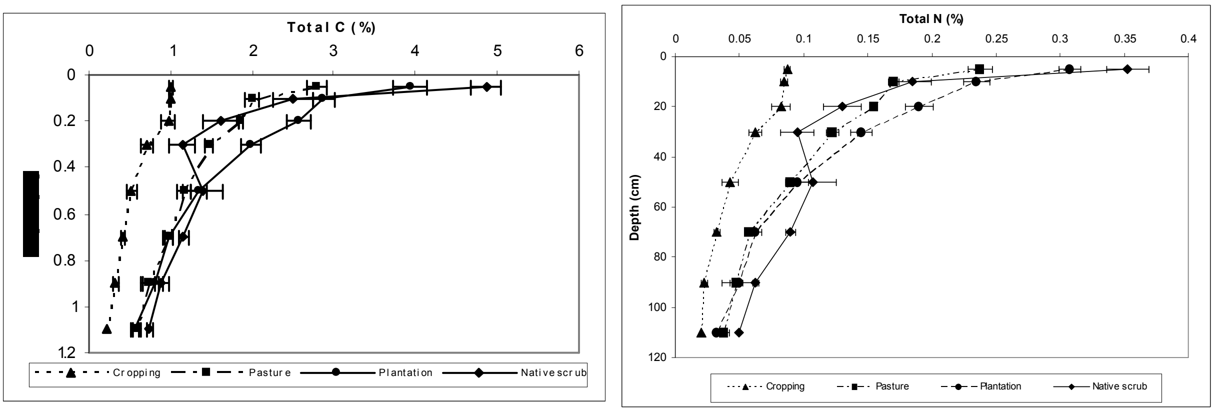

Figure 1. Profile distribution of total C (%) and total N (%) in a Red Ferrosol under four different land uses at Taabinga. Bars indicate standard errors of the means (n=4).

Soil $\delta^{13}C$ was most negative (most depleted in ^{13}C) in the native vine scrub with a value of -25.9‰ in the surface 0.05 m of soil while $\delta^{13}C$ under pasture was -21.7‰ and -22.2‰ under cropping (Figure 2). The $\delta^{13}C$ values at Taabinga under pasture were enriched (or more positive) due to the presence of C_4 grasses that have a $\delta^{13}C$ range of -12 to -18‰, while the peanut crop utilises the C_3 pathway (-24 to -32‰) giving it a lower value than that for pasture. The $\delta^{13}C$ values of the LFC under pasture showed significant ^{13}C enrichment throughout the soil profile (Figure 2) compared to the native scrub, whereas under the spotted gum plantation, significant enrichment of ^{13}C of LFC occurred mostly in the top 0.3 m depths, where C_3 carbon from the spotted gums would be mixing with the C_4 carbon from the pasture that the spotted gums were planted on the pasture soil.

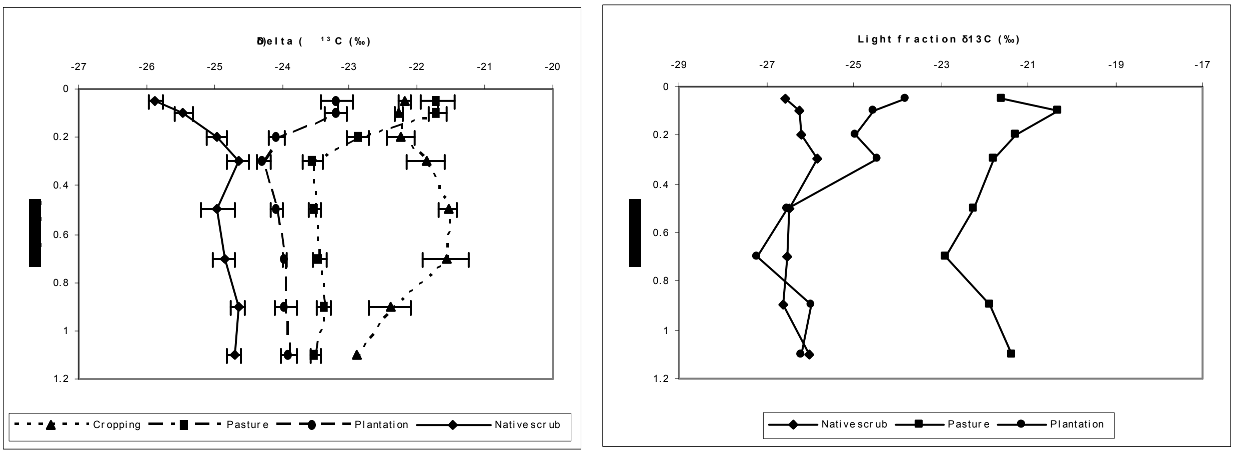

Figure 2. Natural abundance $\delta^{13}C$ (‰) of soil total organic C and light fraction C down the soil profile under different land uses at Taabinga. Bars indicate standard errors of the means (n=4).

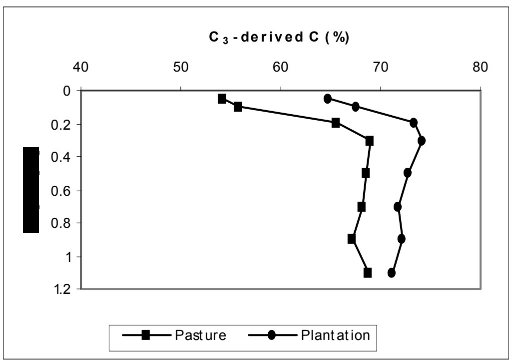

Figure 3. Proportion of C_3-derived C in soil under 4-year-old plantation as compared to adjacent pasture having C_3-derived C from native scrub.

Figure 3 indicates the proportion of C_3-derived C in the 4-year-old spotted gum plantation compared to the adjacent pasture at all depths. In the 0-0.05 m depth, the pasture soil still retains 54% of its C from the previous C_3 sources, while the soil under plantation has increased C_3 source C from 54% to 65%. Thus, in 4 years, the plantation has increased its C_3-derived C by at least 10.6% in the 0-0.05 m soil depth (Figure 3); although at the 1.1 m soil depth there is only a 2.6% increase in C_3-derived C.

Conclusion

This study has demonstrated the major impact of cropping and pasture on decreasing the soil C and N concentrations to a depth of 1.1 m compared to the native vine scrub soil in the Burnett region of southeast Queensland, Australia. Many studies have reported a decrease in soil C and N with afforestation of ex-pasture, but this study indicates that soil C and N may increase after afforestation of ex-pasture land with hardwoods in subtropical Australia. Thus, there is considerable potential for C sequestration in soil under hardwood forest plantations in southeast Queensland and hence, carbon trading incentives. Natural abundance ^{13}C isotope analysis of these four adjacent soils indicated that differences in soil TOC due to the introduction of crop and pasture species were evident down the soil profile. However, the largest differences occurred in the top 0-0.05 m soil depth; with a 10.6% increase in C_3-derived soil C four years after hardwoods were planted onto the pasture soil. A portion of this C_3-derived C from the original native vine scrub was replaced by C_4-derived C from the pasture species and maize crops in the pasture and cropping soils, respectively.

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Biomass carbon: litter quality and implications for carbon sequestration by agroforestry in coastal Kenya

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Abstract

The potential impact of agroforestry systems on C sequestration is widely recognized. However limited data is available on its impact on C dynamics, as much of the previous research has been focused on agricultural productivity. This research was therefore conducted to determine biomass carbon and litter quality of commonly recommended improved fallow species, at the coastal region in Kenya. An on-farm experiment was established in 2006 at a coastal Kenya site, using a Randomized Complete Block Design with five agroforestry species replicated thrice. Biomass carbon and litter quality was assessed at 6 and 12 months after sowing (MAS). Data was analysed using R version 2.9.0. The effects of species, measurement time and their interaction on total carbon were highly significant ($P < 0.001$). Total carbon stocks at 6 MAS ranged from 0.04 (*S.sesban*) to 1.4 Mg C /ha (*M.pruriens*) compared to 1.7 (*S.sesban*) to 20.3 Mg C /ha (*T.candida*) at 12 MAS. Results indicate that substantial amounts of C were sequestered in the biomass. However, duration of the fallow is a key factor to be considered. Litter quality was high with average polyphenol contents of 2.6 % while lignin contents were highest in *T.candida* (16.3 %).

Key Words

Carbon sequestration, agroforestry, litter quality, lignin.

Introduction

The significance of agroforestry with regards to C sequestration has been widely recognized with an estimated global potential of between 12 and 228 Mg/ha (Albrecht *et al.* 2003). However, variability can be high within various agroforestry systems as biomass C stock depends on several factors including environmental conditions, soil type, magnitude of land degradation and the length of fallow period (Albrecht *et al.* 2003; Kaonga *et al.* 2009). Residue quality differences among agroforestry species further play a key role in regulating long term C build up, as the rate of soil organic matter decomposition is dependent on residue chemical quality which is mainly defined using various ratios of carbon, nitrogen, lignin and polyphenols (Vanlauwe *et al.* 1997).

Contrary to the argument that agroforestry systems can only contribute substantial C sinks if the rotation of trees is greater than 20 years, research has shown that short fallows of even less than 5 years represent a substantial C pool (Schroeder 1994; Albrecht and Kandji 2003). However, there is limited data available on impact of C dynamics by agroforestry species, as previous research has focused on agricultural productivity. Therefore, knowing the sizes of carbon pools in agroforestry systems is important to promoting the land use system as a C sink. The objective of this research was to determine the biomass carbon and litter quality of commonly recommended improved fallow species, in the coastal region of Kenya.

Methods

Site description and soil type

An on farm field experiment was established at Malindi Kenya (0^0 12'S 40^0 05E), at an altitude of 20 m ASL, with a mean annual temperature of 25^0C (Njarui *et al.* 2004). The site receives a mean annual rainfall of 1050 mm. The soils are weakly developed Arenosols characterized by a sandy texture with less than 15% clay (Walela *et al.* 2006; MOA 1982).

Field experiment and measurements

The experiment was arranged in a randomized complete block design with five treatments and three replicates during May/June 2006 long rains. The plot sizes were 5m by 5m and treatments comprised of: *Crotolaria grahamiana, Sesbania sesban, Tephrosia vogelii, Mucuna pruriens,* and *Tephrosia candida.*

Above ground biomass production (foliage and woody parts) was assessed at two intervals; 6 and 12 months after sowing (MAS). Aboveground total carbon stocks in the improved fallow species were calculated by multiplying aboveground biomass by a factor of 0.48 (Kaonga 2005). Root biomass was estimated on the basis of a factor of 0.25 of aboveground vegetation biomass (Snowdon *et al.* 2000). Root carbon stocks were calculated by a default conversion factor of 0.26 of aboveground tree C stocks (IPCC 2003). Plant foliage was harvested, oven-dried at 70^{o} C for 48 hours and ground (20 mesh) for plant tissue analysis. Lignin content was determined following the acid detergent fiber (ADF) method while total extractable polyphenols were analysed calorimetrically using the Folin-Cio Calteu method (Anderson and Ingram 1993). Data was analysed using R version 2.9.0 (R Development Core Team 2009). Analysis of variance (ANOVA) was used to determine treatment differences in biomass carbon and substrate quality of improved fallow species.

Results and discussion

Carbon stocks in plant biomass

The effects of species, measurement time and their interaction on total carbon were highly significant ($P < 0.001$). Total carbon stocks at 6 MAS ranged from 0.04 (*S. sesban*) to 1.4 Mg C /ha (*M. pruriens*) compared from 1.7 (*S. sesban*) to 20.3 Mg C /ha (*T. candida*) at 12 MAS (Figure1). Within the individual fallow species, duration of the fallow period significantly ($P < 0.001$) affected the amount of total carbon stocks. The highest increase of 19.3Mg C /ha was observed in *T.candida* with C stocks of between 1.0 to 20.3 Mg C /ha at 6 and 12 MAS respectively. Similar results of *T.candida* accumulating the largest aboveground C stocks as compared to eight other species have been reported in eastern Zambia (Kaonga 2009).

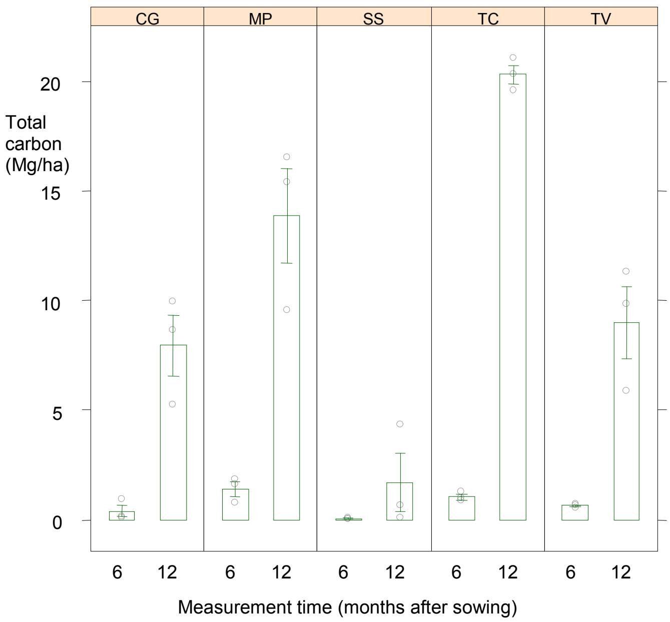

Figure 4. Effect of improved fallow species and time of measurement on total carbon stock accumulation (CG = *Crotolaria grahamiana*, MP = *Mucuna pruriens*, SS=*Sesbania sesban*, TC=*Tephrosia candida*, TV=*Tephrosia vogelii*).

By contrast, C stocks in this study compare similarly with those from other parts of the country particularly Western Kenya. Total C stocks were calculated from published biomass data (Boye 2000; Ndufa 2001; Nybert 2001; Impala 2001) in a 12-month old fallow. The results showed total carbon stocks in *C. grahamiana, S. sesban* and *T. vogelii* were 5.4, 10.3 and 7.1 Mg C /ha, respectively, while the C stocks for the same species in the current research were 7.9, 1.7 and 9.0 Mg C /ha, respectively. These results show similar C biomass build up except for *S. sesban* species. The disparity in performance of *S. sesban* between the two sites can be attributed to variability of climate and edaphic conditions.

Carbon build up in individual fallow species is clearly influenced by the duration of the fallow. A similar trend in C accumulation with a prolonged fallow duration has been calculated from biomass data of a 18-month fallow for *C. grahamiana* and *T.candida* (data from Boye 2000; Ndufa 2001; Nybert 2001; Impala 2001). Calculated C stocks show an accumulation of 17 and 30 Mg C /ha for the two above mentioned species, respectively. The results from the current and related research indicate the need to consider the duration of the fallow species, especially where C sequestration is a major objective. Results from this study would suggest that poorly degraded soils with carbon contents as low as 0.40 %, as those described in the current site (Walela *et al.* 2006), have the potential to sequester substantial amounts of C stocks.

Litter quality
The average polyphenol contents across the agroforestry species were 2.6 %. Lignin contents were highest in *T. candida* (16.3 %) and lowest in *S. sesban* (11.1 %). The lignin to N ratio ranged from 2.80 (*S. sesban*) to 4.52 (*T. candida*) (Table 1).

Table 1. Chemical indices of improved fallow foliage at 6 MAS.

Species	Polyphenols (%)	Lignins (%)	Nitrogen (%)	Ligin:Nitrogen ratio
Crotolaria grahamiana	2.76	11.4	4.03	2.82
Mucuna pruriens	2.37	15.7	5.20	3.02
Sesbania sesban	2.59	11.1	3.97	2.80
Tephrosia candida	2.87	16.3	3.60	4.52
Tephrosia vogelii	2.53	16.1	4.29	3.75

The quality of these species are similar to those reported from published data on agroforestry species; N > 2.5 %, lignin < 15% and polyphenols < 4% (Palm *et al.* 2001). However, *Tephrosia* provenances had a significantly slightly higher lignin content of > 15 % and the lignin to N ratio was also slightly higher in these species. In terms of building recalcitrant C pools in the soil, species with a slightly higher lignin to N ratio would be more effective. Lignin compounds exhibit a higher resistance to microbial degradation and hence are very important for potential of C sequestration.

Conclusions

Total biomass carbon in the improved fallow species tested at the coastal Kenya site increased in the order of *T.candida > M.pruriens > C.grahamiana > T.vogelii > S.sesban.* Further research on optimal duration for maximum biomass C sequestration for agroforestry species in specific climatic zones is needed. The litter quality of the species reported here is high; however, further detailed research is required to determine how litter quality affects the transformations of plant residues into stable soil organic matter. Additionally, the contribution of total biomass carbon of the species evaluated to total soil organic carbon and pool sizes will need to be determined.

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“Black is the new green”: the blue shades of biochar

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Abstract
In recognition of the potential role that “Biochar or Black Carbon” can play in sequestration of carbon, reducing the emission of greenhouse gases and improving the soil fertility, phrases such as “Black is the New Green” have been coined (Marris 2006) and the International Biochar Initiative (www.biochar-international.org) has been started. While the benefits of biochar applications to soil fertility have been well recognised, the potential negative implications of biochar amendment to soils, especially the impact on contaminant fate, dispersal and build-up is thus far not fully appreciated. This paper presents research that shows that biochar addition to soil can potentially lead to accumulation of contaminants residues in soil. The highly reactive biochar can render applied pesticide ineffective and consequently much higher application rates may be needed for the desired pest and disease control. Biochar itself can potentially serve as a source of combustion related toxicants such as polynuclear aromatic hydrocarbons (PAHs) and dioxins. The implication of biochar amendment to soil and potential implications for the environmental accumulation, distribution and food safety of pesticides needs to be fully understood before recommending widespread application of biochar to soils as a climate change mitigation initiative.

Key Words
Biochar, carbon sequestration, climate change, contaminants, pesticides, efficacy.

Introduction
The importance of biochar in sequestration of carbon, reducing the emission of greenhouse gases and improving the soil fertility has led to the International Biochar Initiative (www.biochar-international.org) promoting biochar as a soil amendment which is increasingly attracting the attention of policy makers in USA (Bracmort 2009), Australia and elsewhere. “Black is the new green” as one of the articles in Nature (Marris 2006) put it. Benefits of biochar applications to soil have been well recognised and articulated. However, there are some potential negative implications of biochar application to soils especially the impact on contaminant dynamics in soil environment. The long term impact of biochar amendments on environment to soils is yet to be fully understood.

Biochar has been shown to be particularly effective in sorption and sequestration of organic contaminants in soil due to its greater surface area, high microporosity and other physiochemical properties (e.g. Accardi-Dey and Gschwend 2003; Chun *et al.* 2004; Yu *et al.* 2006). Biochars produced from burning of wheat and rice residues were reported to be up to 2500 times more effective in sorbing organic contaminants than soil (Yang and Sheng 2003, James *et al.* 2005). In a previous study, Yu *et al.* (2006) reported that the sorption and desorption behaviour of diuron herbicide is strongly influenced by the presence of biochars in soil. While this may be a desirable outcome in managing a contaminated soil, the strong affinity of biochar for organic contaminants may also have a downside in agricultural soil. The presence of small amounts of biochar in soils can rapidly inactivate the applied amount of pesticide to soil thus rendering it ineffective and potentially requiring much higher rates of applications of pesticides inputs. Besides, the biochar could itself serve as a source of dispersal of toxic organic contaminants such as PAHs and dioxins, which are produced during the combustion process itself. The objective of this study was therefore to assess the impact of biochar amendment to soil on degradation of applied pesticide as well as its effect on plant uptake of pesticides and potential implication for the efficacy of pesticides in soil.

Methods
Biochars
The biochars were produced from Red gum wood (*Eucalyptus* spp.) at two different temperatures (450 and 850 ^{o}C) as described previously (Yu *et al.* 2006). The woodchips were pyrolyzed at 450 and 850 ^{o}C under limited oxygen in a muffle furnace to make two types of biochars (referred to as BC450 and BC850). The

specific surface area (SSA) of BC850 and BC450 were 566 m^2/g and 27 m^2/g, respectively. BC850 was a microporous material with all pores being essentially less than 1 nm in pore width and the maximum peak occurring at pore widths of about 0.49 nm, whereas BC450 had a lower level of microporosity with the peak maxima occurring at a pore width of about 1.1 nm (Yu *et al.* 2006).

Soil and pesticides
A red-brown earth (a Xeralf) was collected from the Roseworthy Campus of the University of Adelaide. The soil consisted of 87.8% sand, 1.3% silt, 8.3% clay and 1.4% organic matter. The soil had a pH of 6.8 (1:5, soil:water), a maximum water holding capacity (MWHC) of 35% (v/v) and a cation exchange capacity of 9.3 $cmol_c$ /kg. After air drying, the soil was passed through a 2 mm sieve. Biochar amended soils were prepared by thoroughly mixing the soil with accurately weighed biochar on a rotary shaker for 7 d. The percentages of two biochar materials in the amended soils were: 0, 0.1, 0.5, and 1.0% (w/w), respectively.

Two insecticides (carbofuran and chlorpyrifos) were selected in this study because these are widely used to control soil insect pests and their residues have been found in some vegetables in China (Yu *et al.* 2006). Carbofuran is a nonvolatile carbamate compound with a vapour pressure of 0.031 mPa at 20 ^{o}C, a water solubility of 320 mg/L at 20 ^{o}C, and a log K_{ow} of 1.52. Chlorpyrifos is an organophosphate pesticide with a low water solubility (1.4 mg/L at 25 ^{o}C) and high hydrophobicity (log K_{ow} of 4.70).

Plant experiment.
Spring onion (*Allium cepa*) planted in vermiculite was used as the test plant in this study. Seedlings of about 20 cm in height were planted in plastic containers (10 cm in diameter and 10 cm in height) as a closed system allowing no leaching loss of water or pesticides. The soil (500 g) in each container was spiked at concentration of 50 mg/kg for each of the two pesticides. The seven biochar amendments used in this experiment were control (0% biochar), three amendments with BC450 (0.1, 0.5 and 1.0% BC450) and the other three with BC850 (0.1, 0.5 and 1.0% BC850). Each treatment was carried out in five replicates. The amended soils were thoroughly mixed and shaken for 24 h in a rotary shaker, which was followed by evaporation in acetone for another 2 d. Sufficient water was added into each container to adjust the content of water in the soils to about 50% of maximum water holding capacity. An aliquot of 5 g soil was taken out from each container to determine the initial pesticide concentrations. The growth chamber was maintained at 28/20^{o}C day/night temperatures with a 12 h lighting cycle. The plants were watered every 2 d to maintain the soil moisture.

Residue analysis
Five weeks after planting, the plants were cut at the soil level and weighed to obtain the fresh weights of the above-ground biomass. The underground parts of the plants were removed from the substrate and thoroughly washed with tap water to remove the substrate on the surface of the roots, then air-dried at the room temperature for 24 h. The underground parts were also weighed to obtain the fresh biomass weights. After all the plants were removed, a small quantity (5 g) of the soil samples was collected for analysis after thorough mixing. An aliquot of 5 g plant sample was mixed with 20 g of sodium sulphate dehydrate and ground in mortar and pestle. The mixtures were then extracted with 30 mL of solvent (acetone/hexane (1:1, v/v) for chlorpyrifos, and acetone for carbofuran). For both pesticides, the amounts of residues in soils were analyzed by HPLC. However, the residues of chlorpyrifos in plant samples were analyzed by GC-MS and of carbofuran by HPLC. Further details on analytical method have been published elsewhere (Yu *et al.* 2009).

Results and Discussion

Decreased bioavailability to microbes and increased persistence with increasing biochar content in soil
Persistence of both carbofuran and chlorpyrifos insecticides increased with increased biochar content in soil indicating reduced bioavailability to soil microorganisms, as shown in Figure 1 for carbofuran as an example. At the end of 35 d of incubation, a total of 86% of applied chlorpyrifos and 88% of carbofuran residues were lost from the control treatment, whereas only 44% chlorpyrifos and 51% of carbofuran degraded from the soil amended with 1.0% BC850. BC450 inhibited the rate of decay to a much lesser degree. Similar results showing increased persistence of diuron and benzonitrile by selected microorganisms in the presence of wheat char have been reported by other workers (Zhang *et al.* 2005; Yang *et al.* 2006).

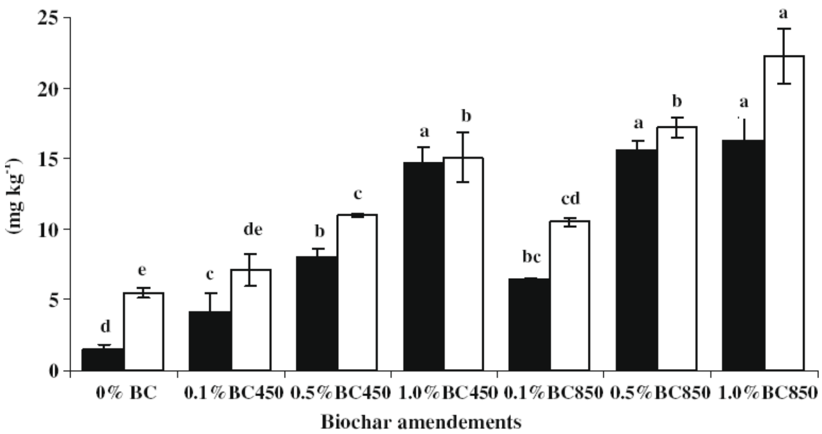

Figure 1. Demonstration of increased persistence of carbofuran insecticide in soil with increasing contents of two types of biochar in soil (BC450 and BC850 are two wood biochars produced at 450° C and 850°C, respectively). Different letters above the same bar type indicate significant difference (Duncan, $p < 0.05$). Source: Yu *et al.* 2009.

Decreased phyto-availability and plant uptake in the presence of biochar in soil

The data in Figure 2 show that the residues in both above-ground parts as well as below-ground parts of spring onions for both pesticides progressively decreased in the plants that were grown in soils amended with increasing amounts of biochars, especially that produced at higher temperature (BC 850). For example, the concentration of carbofuran in the underground plant parts decreased from 14.4±0.8 in control soil to only 1.8±0.4 mg/kg in the soils amended with 1.0% BC850. Similarly the corresponding decrease of chlorpyrifos uptake was from 14.1±1.7 to 0.8±0.1 mg/kg in the presence of 1% BC850. The residues in the above-ground parts were found to be 20-250 times lower than those in the underground parts for both pesticides (Figure 2). The residues of carbofuran were generally higher than those for chlropyrifos for any treatment, presumably due to the lower hydrophobicity of the former.

Clearly the bioavailability of pesticides to microbes for degradation (Figure 1) as well as to plants for uptake decreased as the content of the biochars in soil increased (Figure 2). This shows that the efficacy of applied pesticides could be markedly reduced in the presence of highly reactive biochars, such as BC850 used in this study. Incorporation of small amounts of biochar in a soil has also been noted to inhibit the microbial degradation of pesticides and reduce herbicidal efficacy by other workers also (e.g. Yang *et al.* 2006).

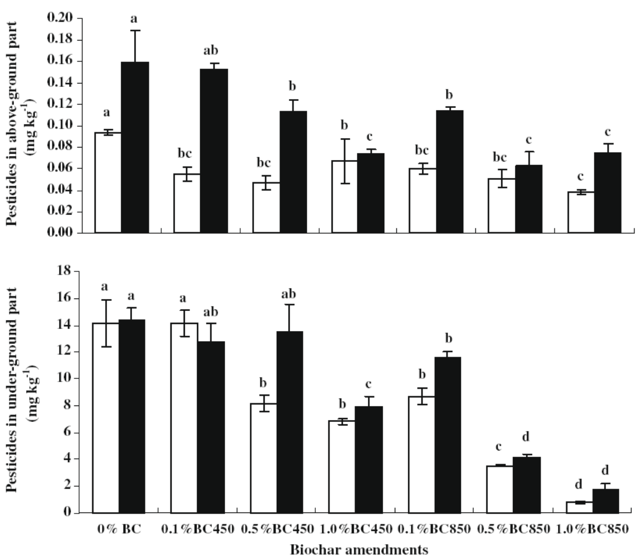

Figure 2. Demonstration of decreased plant uptake of two insecticides (carbofuran – solid bars and chlorpyrifos – empty bars) in above- and under-ground plant parts of spring onion with increasing contents of two types of biochars in soil (BC450 and BC850 are two wood biochars produced at 450° C and 850°C, respectively). Different letters above the same bar type indicate significant difference (Duncan, $p < 0.05$). Source: Yu *et al.* 2009.

Conclusion and Implications

The data presented above show that biochar amendment to soil can potentially lead to accumulation of contaminants residues in soil. Incorporation of a small amount of biochar in a soil has also been shown to inhibit the microbial degradation of pesticides and reduce herbicidal efficacy by other workers. The highly reactive biochar can render the applied pesticides ineffective and much higher pesticide application rates may be needed for the desired control of pests and diseases. Biochar itself can potential serve as a source of combustion related contaminants such as polynuclear aromatic hydrocarbons (PAHs) and dioxins. The implications for long term fate of contaminants and efficacy of applied pesticides in controlling pest and diseases need to be established before adopting the practice of widespread application of biochar as a climate change mitigation initiative.

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Building soil carbon content of Texas Vertisols

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Abstract
Soils in central Texas, USA (Udic and Entic Pellusterts) have been degraded by intensive agricultural practices. This was typified by loss of soil organic carbon from about 1880 to 1949 from a concentration of about 6.5% in the surface to about 1%. Agriculture practices since 1949 are slowly rebuilding the soil carbon content. Recent research has shown that modern conventional practices have increased soil organic carbon sequestration at a rate of 0.15 Mg C/ha/yr. Intensive management practices such as no-tillage increase this rate an additional 0.3 Mg C/ha/yr. Conversion from row cropping to perennial grass production increases sequestration to 0.45 Mg C/ha/yr. Several management options are available to sequester carbon in central Texas soils.

Key Words
Tillage, grass establishment, no-till

Introduction
Blackland soils in central Texas, USA consist primarily of 4.5 million hectares of Vertisols (Udic and Entic Pellusterts) (Puentes *et al.* 1988). These deep and dark colored soils were the basis for much of Texas early agriculture. Also, many of Texas major cities have developed in the Blackland region. The central Texas region was used for free range cattle grazing for the most part until about 1875 when the railroads began to extend into the region. Cotton (*Gossypium hirsutum)* production increased greatly at that time and, by 1909, almost 1/3 of the cotton in the world was being produced in this area (Paddock, 1911). Other common crops at that time included corn *(Zea mays)* and hay. By the 1920s more than 70% of the Blackland Vertisols were tilled to produce crops, using the inversion tillage practices common at that time (US Department of Agriculture 1993). Early tillage practices in this region were intensive and involved moldboard plowing as the primary practice to breakout the field from the original prairie vegetation. The fields were commonly tilled eight to ten times during the growing season to prepare the seed bed and to control weeds. A typical management schedule included moldboard plowing, several passes with a disk, harrow, plant, and three or more cultivations before harvest. In retrospect, a benefit of intensive tillage was to oxidize organic matter in the soil and thus release plant nutrients for the growing crop. By the early 1940s, a common rotation was two years of cotton and three years of small grain (Richardson 1993). Cotton produces small amounts of residue returned to the soil to replenish the organic carbon oxidized by the agricultural practices of the time. As a result of the agricultural practices commonly used at that time and the choice of crops, tilled soils were severely depleted of organic carbon compared to the native prairie in 1949.

Modern agricultural practices have changed greatly from the past and have produced changes in soil carbon content. About 80% of the Blackland Vertisols are in farms and ranches. Of that amount, about half are planted to croplands and the rest predominately are improved pasture. Wheat *(Triticum aestivum L.),* oat *(Avena sativa),* grain sorghum *(Sorghum bicolor*)*,* corn *(Zea mays)* and cotton (*Gossypium hirsutum)* are the major crops in the region at present. Only very small amounts of native grassland still exist. An opportunity to compare soil properties with a known starting condition was discovered recently by the USDA-ARS Grassland Soil and Water Research Laboratory for a study area near Riesel, Texas. In 1949, a series of soil samples were taken from five fields at the GSWRL-Riesel site (Baird, 1950). The samples were oven dried and placed in labeled cartons and stored in a dry location for over 55 years. The management history of these fields was recorded for most of the intervening years. The goal of the original study was to determine the effect of cropping on soil water storage. In 2004, these fields were sampled again. The objective of the current study was to compare soil properties between the management regimes and between the two sampling periods. This report deals with the soil organic carbon content comparisons of the archived and modern soil samples. Results will be compared to other recent findings for selected management practices.

Methods
Soil samples were taken to a depth greater than 90 cm with a hand auger in 1949 and hydraulic core tubes in recent studies. Cores were divided into sections representing depth increments of 0 to 5, 5 to 10, 10 to 15, 15 to 20, 20 to 30, 30 to 60, 60 to 90, and 90 to 120 cm depth. Soil water content was determined on an aliquot

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and bulk densities were determined for modern samples. The remainder of the segment was gently crumbled, passed through a 2mm sieve and air dried. For all samplings, plant stem and root segments were hand picked from the soil samples. The samples were ground in a ball mill to pass through a 250-µm sieve and then stored at room temperature in glass bottles. A sample aliquot (approximately 1 g) was analyzed for organic carbon with a Leco CR412 Carbon Determinator (Leco Corp, Augusta, GA, USA) using the combustion method of Chichester and Chaison (1992). While carbonates were present in the soils, this method of analysis differentiates between organic and inorganic carbon by burning the organic carbon at a temperature of 500 ºC that leaves the inorganic carbon relatively intact. The inorganic carbon was later determined by burning at 1050 ºC to obtain the total carbon and subtracting the amount of organic carbon. Carbon mass in the segment was determined by calculating the amount of carbon in a depth increment by multiplying the carbon concentration by the bulk density. Statistical analysis was conducted using t-tests to determine differences in soil carbon content.

Results

The soils studied in this project are located near Riesel and Temple in central Texas, USA. Mean annual temperature is 19.5 °C and average rainfall ranges from 840 to 908 mm/yr. Soils in all fields are Vertisols (Udic Pellusterts) (Soil Survey Division Staff 1993).

A concern with this study was the possibility that some of the archived samples soil organic carbon had been lost while in storage. The results from the native prairie sites were from both sampling periods were compared to see if differences occurred with similar management. Soil organic carbon concentration was significantly greater in the surface 15 cm for the 2004 sampling period than in the 1949 sampling period, 2.77 percent in 1949 and 3.31 percent in 2004. However, mean differences in concentration between the two sampling periods were not significantly different for depths greater than 15 cm. Prior to 1949, the native prairie site had been grazed for an extended period of time.

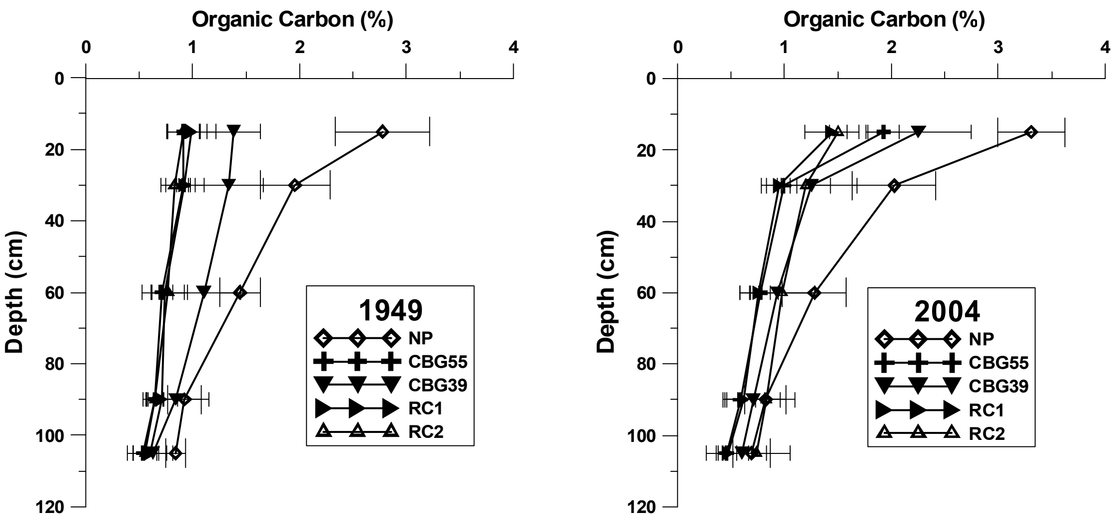

Figure 1. Comparisons of soil organic carbon profiles from 1949 and 2004 samples for selected management practices. NP is Native Prairie, CBG is Coastal Bermuda Grass, and RC is Row Crop. Error bars represent ± one standard deviation. (From Potter 2006).

Compared to native grasslands, the soil organic carbon was greatly depleted by agricultural practices prior to 1949. Soil organic carbon concentrations in the tilled fields were greatly reduced compared to that found in the Native Prairie (Figure 1). T-tests comparing the organic carbon concentration of NP to the other soils revealed significant differences ($P<0.05$) in concentration to a depth of at least 90 cm. Organic carbon concentration had been reduced throughout the profile to a depth of 90 cm by early agricultural practices. The mean organic carbon concentration in the 1949 soil samples was similar throughout the profile with a mean concentration of about 1%.

Differences in soil organic carbon between the 1949 samples and the 2004 samples are a result of management and weather effects. In the 1949 time frame, farming practices changed dramatically. One major change was the increased use of fertilizers (US Department of Agriculture 1993). At the same time, plant populations also increased dramatically with the improvement of crop varieties. Cotton production was greatly reduced on the site. This has resulted in a large increase in the amount of residue being returned to the soil, which in turn replenished some of the organic matter being oxidized.

Soils in the two fields with nearly continuous crop production (RC1 and RC2) differed slightly in the soil organic carbon profiles between sampling periods. Predominant crops during this period included corn, wheat and grain sorghum. Soils in field RC1 were very similar to the soils in the 1949 sampling, with the exception of a significant increase in carbon concentration from 0 to 15 cm in 2004. Soils in RC2 from the 2004 sampling period also had a larger organic carbon concentration in the surface soils, which extended to 30 cm in this field. Below 30 cm, carbon concentrations were similar in 1949 and 2004. Estimating the amount of C sequestered by modern farming methods using the difference between the 2004 samples and 1949 samples, soils in field RC1 sequestered 8.7 Mg C/ha (158 kg C/ha/yr) and soils in field RC2 sequestered 6.9 Mg C/ha (125 kg C/ha /yr) in the surface 30 cm during the 55 year time interval.

It should be noted that the change in soil carbon concentration found in this study is without the use of no-tillage management practices. In a study comparing no-till with conventional chisel plow management, after 10-yrs continuous management additional differences in soil organic carbon concentration were found in the surface 20 cm of soil (Table 1) (Potter *et al.* 1998). After a corn crop, differences in organic carbon concentration were greatest in the surface 4 cm. No-till resulted in an annual increase in soil carbon of 0.3 Mg C/ha/yr greater than that found in a chisel plow conventional tillage soil.

Table 1. Carbon sequestration rates for Vertisols in central Texas with selected management practices.

Management	Sequestration rate Mg C/ha/yr	Information source
Conventional till	0.12	Potter 2006
No-till	0.30	Potter *et al.* 1998
Perennial grass	0.45	Potter *et al.* 1999

The soil in the two fields that had been planted to perennial Coastal Bermuda Grass (*Cynodon dactylon* (L.) Pers.) for 55 and 39 years, showed similar trends with an increase in soil organic carbon near the surface. Below 60 cm the differences between the 1949 and 2004 samples were not significantly different for these two fields. Restoring the soils to perennial grass vegetation replenished the carbon concentration in the surface.

The estimated amount of carbon sequestered depends on the amount of carbon stored in the soil at the start of the study. To calculate the amount of carbon stored in the soil by taking the difference between the mean 2004 soil carbon and the mean 1949 soil carbon, It was assumed that the soil bulk densities were the same in both sampling periods. Soils in the 39-year grass field sequestered 13.5 Mg/ha C in the surface 30 cm, while in the 55-year grass field, soils sequestered 19.7 Mg C/ha in the surface 30 cm. If, however, the 1949 samples had greater bulk density values than the 2004 samples, then the estimated amounts of carbon sequestered would be less. If the 1949 bulk density values were lower, then estimated amounts of carbon sequestered would be greater. Without measured values, it is difficult to be certain of the 1949 bulk density values. Larger soil organic matter content in 2004 would tend to reduce bulk density in the recent samples. Conversely, the lighter equipment used in 1949 may have resulted in lower bulk density when the archived soils were sampled. Given the assumption of similar bulk density values, carbon sequestration appears to increase throughout the time that grass is grown which agrees with previous work (Potter *et al.* 1999). On similar soils, grass established for periods from 6 to 60 years sequestered carbon in a linear fashion and at a mean rate of 0.45 Mg C/ha/yr (Figure 2) (Potter *et al.* 1999).

Conclusion

Soils in central Texas were degraded during the first 70 years of cultivation. This is illustrated by the loss of soil organic carbon. In the last 60 years changes in crops, fertilization and management practices has resulted in a slow rebuilding of the soil organic carbon in the soil profile. Use of no-tillage and conversion to perennial grass accelerates the accumulation of soil carbon.

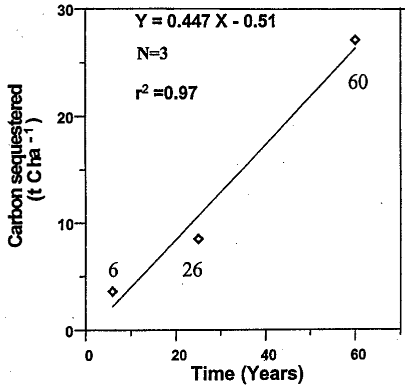

Figure 2. Carbon sequestered in soil after periods of 6, 26, and 60 years of grass establishment after extensive periods of cultivation (from Potter *et al.* 1999).

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Carbon Sources and Dynamics in Afforested and Cultivated US Corn Belt Soils

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Abstract
Afforestation of degraded cropland can sequester atmospheric carbon (C), but soil organic C (SOC) sources in such ecosystems are not well-characterized. This study assessed SOC dynamics and sources in two 35-yr-old, coniferous afforestation sites [i.e., a forest plantation and a shelterbelt situated at northwestern Iowa (Sac) and eastern Nebraska (Mead), respectively] and the adjacent agricultural fields. Composite soil samples were collected at both sites to determine OC and total nitrogen (TN) contents, and stable C isotope ratios ($\delta^{13}C$, natural abundance). In these fine-textured soils, afforestation of cropland carried out through either shelterbelt or forest plantation caused substantial increases in SOC accrual (≥ 57%; $P < 0.05$) in surface soil layers (to 7.5 or 10 cm deep) relative to conventionally-tilled cropping systems. Soils exhibited a marked gradient of $\delta^{13}C$ signatures from near constant values in cropped fields (-17 ± 0.1‰) to much depleted in afforested soils (-22 ± 0.4‰) indicating a gradual shift in C sources. Source-partitioning assessments revealed that tree-derived C contributed roughly half of the SOC found directly beneath trees indicating that the additional SOC accrued in these afforested sites can be fully explained by tree-derived inputs.

Key Words
Land-use system, management choice, soil quality, soil resilience, ecosystem services.

Introduction
Soils can act as a net sink of atmospheric C (Follett *et al*. 1997) creating opportunities to mitigate global climate change. Concomitant increases in *in situ* SOC quantities can concurrently enhance soil quality and overall ecosystem resiliency. These various soil processes can be influenced by vegetation type. Within this context, the US Corn Belt landscape is dominated by corn (*Zea mays* L.) and soybean [*Glycine max* (L.) Merr.] with these two crops covering 74% of total land surface (NASS 2009). There is abundant knowledge currently available about the comparative effects of these common cropping systems on SOC accrual (Huggins *et al*. 1998). Fewer studies have focused on the practice of tree planting in degraded cropland as a means for SOC accretion and soil quality restoration. After assessing a mature shelterbelt in Nebraska, Sauer *et al*. (2007) reported the advantage of afforestation over cultivation for SOC accrual. Likewise, evaluating two locations in Ohio, Bronick and Lal (2005) found enhanced SOC contents in forest vs. cropland. However, knowledge concerning the relative effects of afforestation vs. cultivation as well as the associated ecophysical factors governing SOC accretion and sources still remains incomplete and fragmented. With the aim of attaining additional insights about SOC dynamics in these ecosystems, conventional SOC inventories need to be supplemented with information about retention of newly-added C into SOC. This critical understanding about SOC dynamics can in part be acquired through assessing C source-partitioning. Using isotope methods, McPherson *et al*. (1993) discriminated the tree-C contribution to SOC in forest-prairie ecotones. Currently, there is no information available about SOC sources for afforested ecosystems in prairie-derived soils. The objective of this study was to assess the relative impacts of cropping vs. afforestation systems on SOC accretion and plant-C sources in two sites within the US Corn Belt.

Materials and Methods
This study was conducted at two sites: Mead and Sac. Mead is located within the University of Nebraska-Lincoln ARDC, NE (41° 9' N, 96° 29' W, 356 m elevation) with soil series Tomek silt loam (fine, smectitic, mesic Pachic Argiudoll) (USDA-NRCS 2002). This site consists of a 35-yr-old, north-south oriented shelterbelt and the two adjacent cultivated fields to the west and east sides of the shelterbelt. Tree species included eastern red cedar (*Juniperus virginiana*) and scotch pine (*Pinus sylvestris*). Trees were arranged in two parallel rows with distances of 3.65 m between rows and 1.8 m between neighbouring trees within rows. The adjacent fields were primarily cultivated to wheat (*Triticum aestivum* L.) –corn–soybean rotation using fall chisel plowing. A rectangular grid for soil sample collection was established across the shelterbelt and the two adjacent cultivated fields with 7 × 17 (north-south × east-west) sampling points distributed in an area of 304.6 m^2. Composite samples (n= 4) were collected near each grid point with 0-7.5 and 7.5-15 cm depth

increments. The Sac site is located at Early, IA (42° 26’ N, 95° 9’ W, 401 m elevation) with soil series Galva silty clay loam (fine-silty, mixed, superactive, mesic Typic Hapludoll) (USDA-NRCS 2002). This site consists of a 35-yr-old eastern white pine (*Pinus strobus* L.) forest plantation (≈ 5.1 ha) and an adjacent, commercial field under corn-soybean rotation. Distance between pine trees (within rows) and between tree rows averaged 2.73 and 3.50 m, respectively. Tree diameter measured at 1.3 m height was 0.25 m and tree height was 14.0 m, respectively. Prior to soil sample collection, the crop field had a long-term history of tillage (≈ 30 yr under chisel plowing). For soil sample collection, Sac afforested and cropped fields were divided into polygons (5×5 m^2), and 5 polygons were randomly selected within each field. Then, 5 composite soil samples (n= 2) were collected within each selected polygon at 0-10, 10-20, and 20-30 cm depth increments. All Mead and Sac soil samples were collected after crop harvest and before fall tillage operation using a 3.2-cm i.d., hammer-driven, split-tube probe after surface plant residue was brushed aside. Leaf and branch samples of dominant tree species at both Mead and Sac sites, and undisturbed soil samples from a native prairie vegetation site near Sac [i.e., Kiowa Area, 42° 28’ N, 95° 6’ W; soil series Clarion loam (fine-loamy, mixed, superactive, mesic Typic Hapludoll] (USDA-NRCS 2002) were also collected as reference materials for SOC sources estimations. Using conventional methods, all samples were dried and ground to powder consistency. We determined OC, TN, and $\delta^{13}C$ isotopic composition via dry combustion method using a Fison NA 15000 Elemental Analyzer interfaced to an isotope-ratio mass spectrometer Delta V Advantage. Because carbonates were presented in Mead soils, a pressure calcimeter method (Sherrod *et al*. 2002) was used to determine and discount soil inorganic C. The SOC mass storage was calculated by multiplying SOC concentration, ρ_b, and soil layer thickness. Gravel was not present in these loess-derived soils. Mass balance for SOC sources was estimated using $\delta^{13}C$ measurements in soil and plant samples as well as reference values by Follett *et al*. (1997). Mean residence time (MRT) of SOC was calculated as Dorodnikov *et al*. (2007). Bulk density (ρ_b) and soil pH (1:1 in water) were also quantified. Analyses of variance (ANOVA) models and Tukey tests at a critical value of 0.05 were run to examine land-use effects.

Results and Discussion

Afforestation Effects on Carbon and Nitrogen Accretion

At the Mead site SOC (19 vs. 38 g/kg; Figure 1A) and TN (1.8 vs. 3.0 g/kg; Figure 1B) contents sharply increase in the surface soil layer ($P < 0.05$) as gradually transitioning from the crop fields to the tree rows. The shallowest soil layer at the Sac site also showed similar results (pine plantation > crop field, $P < 0.001$; Figure 2A; Figure 2B). All three soil layers at the Sac site exhibited consistently wider C/N ratios for the afforested soil ($P < 0.001$; Figure 2C). The divergent C/N ratios between afforested and cropped soils suggest the quality of organic matter differs for these two ecosystems. This result is consistent with data by Martens *et al.* (2003) and Sauer *et al.* (2007) for afforested soils receiving no N fertilizer additions. The absence of exogenous N additions coupled with both a fungal-dominated microbial community beneath trees (Ohtonen *et al.* 1999) and the relatively low quality of C inputs from trees (Melillo *et al.* 1989) could explain these marked patterns of wider C/N ratios in the afforested soils. The SOC expressed on a mass-volume basis for the 0-30 cm profile at Sac indicates greater SOC accrual in the pine-afforested vs. cropped soil (Table 1). A relatively lower SOC amount in the surface depth increment of the annually-plowed, cropped soil suggests that C accrual took place mainly in the surface layer of the pine-afforested soil and/or a tillage-induced C depletion occurred in the surface layer of the cropped soil. Surface soil ρ_b was similar for two Sac fields and roughly 10% higher in the crop field at 10-30 cm depth ($P \leq 0.001$; data not shown). Soil pH was typically lower in afforested vs. cropped soils by 0.6 units (5.6 vs. 6.2) ($P \leq 0.002$; data not shown).

Our finding of enhanced SOC accrual in afforested surface soils is in agreement with previous studies under a broad variety of ecophysical conditions. Bronick and Lal (2005) reported roughly 2-fold increased SOC contents in wooded vs. cultivated land in Ohio. Martens *et al.* (2003) found 46% increases in SOC content in afforested vs. cropped land in Nebraska in association with enhanced soil aggregation. Likewise, examining a grassland-woodland ecotone in Arizona, McPherson *et al*. (1993) observed greater SOC in woodland vs. grassland sites (≈ 21%) to be in close association with increased root biomass in their tree-covered locations. Similarly, after evaluating a chronosequence (1–29 yr) of afforestation in cropland in Denmark, Vesterdal *et al*. (2002) quantified SOC increases with time at the 0-5 cm depth increment. Conversely, they also detected gradual SOC depletion with time deeper in the soil profile (5-25 cm). This outcome apparently contradicts the majority of existing reports as well as our findings; however, this could be in part attributed to a low soil capacity for SOC stabilization and protection caused by a lack of clay mineral surfaces in their coarse-textured soils (i.e., sandy loam, 69% sand particles). Other previous reports also support the hypotheses that soil texture and mineralogy as well as quantity and quality of tree-C inputs are key controlling factors of

SOC accrual in afforested soils (Melillo *et al.* 1989; Richter *et al.* 1999). Additionally, it can be anticipated that continual, large tree litter production, canopy cover, and water uptake at our conifer-afforested locations could have created consistently cold-dry soil conditions likely causing deceleration in residue-SOC decomposition and enhancing SOC accretion compared to conventionally-tilled cropped fields. These and other inherent aspects of land-use conversion from cropland to forest such as tillage cessation and soil erosion reduction could collectively contribute to enhanced C sequestration in afforested fine-textured soils.

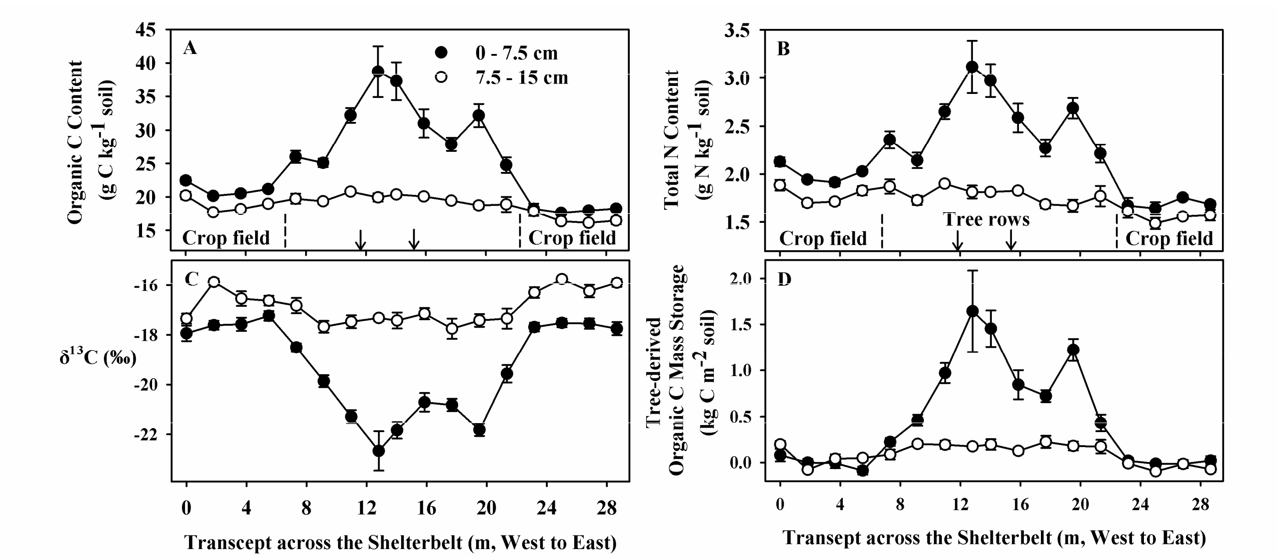

Figure 1. (A) Soil organic carbon and (B) total nitrogen concentrations, (C) stable carbon isotope ratios (δ^{13}C), and (D) estimated organic carbon mass derived from tree input in a transept across the Mead site. Fields boundaries are indicated. Each mean value averaged 7 sampling points. Error bars are ±SE.

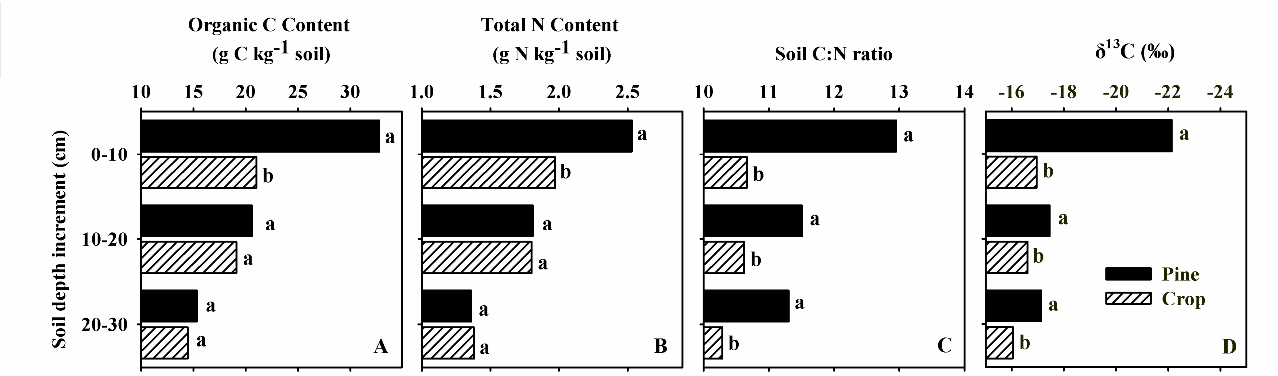

Figure 2. (A) Soil organic carbon and (B) total nitrogen concentrations, (C) organic carbon to total nitrogen (C:N) ratios, and (D) stable carbon isotope ratios (δ^{13}C) at Sac site. Within each depth and variable, land-use systems labelled by the same letter are not different based on Tukey's HSD test (α = 0.05). *n* = 25.

Table 1. Soil organic carbon (SOC) in a forest plantation and adjacent cultivated field at the Sac site, Iowa.

	SOC Mass Storage (Mg/ha)			
	Soil depth, cm			
Treatment or statistic	0 - 10	10 - 20	20 - 30	0 – 30
Eastern white pine (35-yr-old)	33.6	23.5	19.6	76.7
Corn - soybean rotation	22.3	25.4	20.3	68.0
P > *F* (probability after ANOVA models)	<0.001	NS	NS	<0.001

Stable Carbon Isotope Signatures and Afforestation Impacts on Carbon Sources

Mead soils exhibited a marked gradient of δ^{13}C signatures from near constant values in the cropped fields (-17.6 ± 0.1 ‰) to much more depleted between tree rows (-22.3 ± 0.4 ‰) capturing a gradual shift in SOC sources (Figure 3A). Based on mass balance estimations, SOC source-partitioning revealed that tree-derived SOC contributed 54% (i.e., 1.7 ± 0.2 kg C/m^2) of the existing SOC found directly between trees (Figure 3B). At Sac, soil δ^{13}C values at all three soil layers exhibited depletion in pine-afforested vs. cropped soils ($P \leq$ 0.05; Figure 2D). At Sac pine-afforested soils, δ^{13}C results revealed that tree-derived SOC corresponded to

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47, 12, and 2% of the existent SOC at 0-10, 10-20, and 20-30 cm depth increments, respectively; therefore, masses of tree-derived SOC at these soil layers corresponds to 15.8, 2.8, and 0.4 Mg C/ha, respectively.

Our results collectively suggest that tree-derived C inputs can fully account for all the SOC replenishment following conversion from cropland. Source assessments also indicate that tree-derived C contributed roughly one-half of the current SOC found in surface soil beneath trees at both Mead and Sac (Figure 1A and Figure 2A). Such quantitative information confirming tree inputs (i.e., litter and shallow roots) as major SOC sources in afforestation ecosystems has not been previously reported. Likewise, our results also allowed numerical assessment of SOC turnover based on the premise that these surface soils had reached a new equilibrium after 35-yr of tree establishment (Richter *et al*. 1999). These C-enriched afforested surface soils exhibited MRT for SOC on the order of decades (i.e., 45 and 55 yr for Mead and Sac, respectively) indicating that SOC beneath trees is subject to an intermediate dynamics compared to both fast turnover for soils planted to *Miscanthus* × *gigantheus* with MRT of 13 yr (Dorodnikov *et al*. 2007) and relatively slow turnover for corn fields with MRT of 117 yr (Huggins *et al*. 1998).

Conclusions

Land-use change from tilled cropland to conifer-afforestation achieved substantial SOC replenishment in fine-textured surface soils suggesting a positive scenario for soil quality restoration via tree planting in degraded cropland. This on-farm study also supports tree litter on soil surface and shallow tree roots as main sources for enriched SOC. Although afforestation strategies registered a beneficial outcome, underlying mechanisms responsible for SOC sequestration remain unclear. The potential impacts of tree biomass removal (i.e., due to growing interest in biofuel fabrication) on SOC reservoir and dynamics also remains uncertain. Quantitative discrimination of litter- and root-derived tree-C inputs, SOC fractionation, net ecosystem productivity, and soil respiration amounts and sources could aid in elucidating these unknowns.

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Chronosequential alterations of properties of post-agrogenic Chernozems of the Kursk steppe zone of Russia under self-restoration

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Abstract
The focus of this chronosequential study was on the vegetation succession, profile morphology, soil nutrient dynamics, soil organic carbon (SOC) stocks, and dynamics of functionally different SOC pools of post-agrogenic Chernozems under self-restoration of the Kursk steppe zone of Russia. The ages of abundance of the post-agrogenic soils were 8, 19, 37, and 59 years. After 59 years of self-restoration the vegetation succession reached the climax stage with dominance of *Stipa pennata* and *Arrhenathum elatius*. In 59 years self-restoration SOC stores increased in the upper 0.5m from 78.9% to 95.3% of the virgin Chernozem. In the cosequence cation exchange capacity (CEC) increased. No significant changes were found in respect of C/N ratios, pH, exchancheable cations, base saturation, and amounts of plant available nutriens. The investigation in respect of functionally different SOC pools of post-agrogenic Chernozems under self-restoration reveal a dominance of OC in the fine silt and clay fractions (36-68% of SOC). Hence the passive C pool was found to be dominating. The free particulate organic matter (POM) fraction was 1-7% of SOC and the occluded POM fraction was 20-58% of SOC. As also found for the others the fraction of the occluded POM (20–58% of SOC) increased during self-restoration, but the rate of occluded POM after 59 years of self-restoration didn't reach the level of the virgin Chernozem.

Keywords: Self-restoration, Chernozem-chronosequence, soil organic carbon (SOC) stocks, SOC pools, Russia

Introduction
Until recently, much arable land was abandoned in many countries world-wide due to different reasons (wars, economical and ecological crises, intensification of agriculture) (Ramankutty 2006; Lyuri *et al*. 2006). Most abandonment was found in Russia, reaching over 200,000 km^{-2} in the early 1990s (Vuichard *et al*. 2008) and 578,000 km^{-2} in the years 1961-2007 (Lyuri *et al*. 2008). As a consequence, the soils of these abandoned sites went into the process of natural restoration or self-restoration without any direct human impact.

The focus of this chronosequential study was on the vegetation succession, profile morphology, soil chemistry, carbon (SOC) stocks, and dynamics of functionally different SOC pools of post-agrogenic Chernozems under self-restoration of the steppe in the European part of Russia.

Materials and Methods
The study was done in the area of the V.V. Alekhin Central-Chernozem Biosphere State Reserve at 51^0N and 36^0E, which is situated about 18km south of the city of Kursk (Russia). The ages of self-restoration of the post-agrogenic soils were 8, 19, 37, and 59 years. One actual arable soil and one natural soil, never been cultivated, were included in the study as a control. Subsequent sampling sites were chosen according to appropriate information from topographic maps, and personal communications with colleges from the Central-Chernozem Biosphere State Reserve.

Carbonate content, pH, CEC, plant available K and P were determined according to Schlichting *et al*. (1995). The procedure of the physical fractionation to obtain free particulate organic matter (POM), occluded POM and the grain size fractions was done according to Steffens *et al*. (2009). Carbon and nitrogen contents in dry soil pellets were determined after combustion and spectrometric measurements with a C/N/S analyser (CHNS-Analyser Flash EA) as total C and N, within the density fractions $<1.8 g/cm^3$ (free POM and occluded POM) as well as within the grain-size fractions sand, silt, and clay.

Results and Discussion
After 8 years under self-restoration the soil showed the typical crump structure for virgin Chernozem whereas the vegetation changed from crop to a rural appearance of *Agropyron repens*. The former ploughing boundary was still present, also in soil being 19 years in self-restoration, whereas it was not nor visible from

then. After 37 years of self-restoration a species-rich transitional stage in the vegetation succession was achieved, and 59 years after abandonment the vegetation succession reached the climax stage with dominance of *Stipa pennata* and *Arrhenathum elatius*.

After 59 years of self-restoration SOC stores increased in the upper 0.5m from 78.9% to 95.3% of the virgin Chernozem (Figure 1). The chronosequence also showed increasing SOC contents in the upper 0.5m from 33.4 to 43.4g/kg in the mean (Table 1). The increasing of SOC content was especially high in the first 10cm of the soils from 39.2 to 78.2g/kg. Because of concurrent increasing carbon and nitrogen contents in the chronocequence the C/N ratios showed no alterations. Changes in chemical properties were found in respect of CEC, which increased from 33.8 to 39.7cmol/kg during self-restoration. The significant correlation (R^2=0.88) between CEC and SOC content indicated that the increase in CEC resulted from increasing SOM. Beside no changes in soil chemical properties, which comprises pH, carbonate content, exchangeable cations, and plant available phosphorus and potassium, were found.

Table 1. Characteristic chemical properties of the studied soils being 8, 19, 37 and 59 years under self-restoration, of an actual arable land, and of a "Natural" Chernozem, never been cultivated

Depth	C	N	C/N	$pH_{(H2O)}$	$CaCO_3$	Exchangeable cations				Σ	CEC	BS	K_2O	P_2O_5
						Na	K	Mg	Ca					
cm	g kg^{-1}				g kg^{-1}	cmol$_c$ kg^{-1}						%	mg 100g^{-1}	
						Arable land								
10	39.2	2.8	13.8	6.2	0.2	< 0.3	0.33	1.9	18.52	20.8	33.8	61	14.35	2.22
20	38.8	2.7	14.1	6.1	0.2	< 0.3	0.11	2.3	17.56	19.9	33.6	59	10.63	1.23
40	31.4	2.3	13.4	6.4	0.2	< 0.3	0.06	2.5	39954	24.0	29.7	81	7.72	0.27
55	21.0	1.4	14.5	6.9	11.4	< 0.3	0.07	1.8	27.43	29.3	25.5	100	7.98	0.17
86	9.6	0.8	11.4	8.3	14.0	< 0.3	0.07	1.5	29.95	31.5	19.3	100	8.84	0.16
125	9.9	0.5	19.8	8.5	11.2	< 0.3	0.08	1.6	22.81	24.4	16.5	100	nd	nd
						8 years								
10	42.0	2.7	15.6	6.2	2.5	< 0.3	0.37	2.3	18.14	20.8	31.1	67	20.53	1.68
20	39.5	2.5	15.5	6.0	3.2	< 0.3	0.13	2.1	17.3	19.5	30.8	63	9.82	1.27
30	31.2	1.6	19.5	6.6	3.5	< 0.3	0.06	1.9	18.1	20.0	29.7	67	7.69	0.33
55	19.9	1.0	19.0	6.9	3.9	< 0.3	0.07	1.5	16.1	17.6	26.6	66	8.27	0.14
75	12.4	0.3	35.6	6.7	80.7	< 0.3	0.07	1.1	14.2	15.4	23.9	64	6.49	0.06
130	6.4	nf	nd	7.9	141.0	< 0.3	0.08	1.2	22.8	24.1	11.7	100	nd	nd
						19 years								
10	41.6	3.1	13.4	8.0	13.6	< 0.3	0.15	1.4	30.9	32.5	36.0	90	10.40	0.80
20	38.7	2.9	13.4	8.2	20.2	< 0.3	0.07	1.3	28.7	30.0	31.5	95	7.20	0.33
30	29.9	1.9	15.4	8.3	39.1	< 0.3	0.06	1.3	29.7	31.0	29.8	100	6.54	0.12
50	22.2	1.3	16.5	8.4	81.5	< 0.3	0.06	1.3	18.9	20.2	24.6	82	6.05	0.06
80	16.9	0.5	30.8	8.5	118.4	< 0.3	0.05	1.1	14.1	15.3	20.1	76	5.79	0.06
120	13.1	0.3	37.6	8.5	133.9	< 0.3	0.06	1.2	21.9	23.2	16.3	100	5.85	0.06
150	9.2	nf	nd	8.5	132.0	< 0.3	0.06	1.6	34.7	36.3	13.7	100	nd	nd
						37 years								
10	55.0	4.1	13.4	6.6	0.7	< 0.3	0.19	2.5	19.3	22.0	36.2	61	10.60	0.37
20	45.4	3.0	14.4	6.5	0.4	< 0.3	0.08	1.8	15.1	17.0	33.9	50	7.43	0.21
40	39.5	2.5	15.8	6.4	0.3	< 0.3	0.07	1.7	16.3	18.1	32.9	55	7.25	0.16
55	30.8	2.1	14.7	7.0	0.9	< 0.3	0.08	1.4	15.0	16.5	28.5	58	7.89	0.14
80	23.9	1.3	17.7	7.6	3.0	< 0.3	0.08	1.2	13.7	15.0	24.3	62	8.24	0.15
130	21.5	0.5	39.1	8.3	87.8	< 0.3	0.07	0.9	21.6	22.6	16.6	100	6.17	0.06
150	8.4	nf	nd	8.5	122.7	< 0.3	0.08	1.2	28.6	29.9	12.1	100	nd	nd
						59 years								
10	54.7	4.0	13.5	6.2	0.9	< 0.3	0.42	2.3	17.1	19.8	39.7	50	18.84	0.31
20	45.1	3.1	14.5	6.5	0.9	< 0.3	0.11	2.1	19.8	22.0	36.5	60	8.66	0.17
50	39.1	2.6	14.8	6.6	2.7	< 0.3	0.09	1.9	16.7	18,7	32.8	57	8.64	0.14
100	18.6	1.2	15.5	6.8	15.8	< 0.3	0.08	1.4	29.1	30.5	22.8	100	9.48	0.15
130	6.0	nf	nd	8.5	137.0	< 0.3	0.09	1.3	20.1	21.5	15.2	100	nd	nd
						"Natural" Chernozem								
10	78.2	6.1	12.7	6.6	0.8	< 0.3	0.73	3.5	26.1	30.3	44.1	69	26.48	0.61
20	52.6	3.8	13.7	6.6	0.9	< 0.3	0.12	2.7	20.2	23.0	38.0	61	8.04	0.27
60	38.2	2.5	15.1	7.2	2.0	< 0.3	0.1	1.6	17.2	18.9	32.1	59	7.58	0.18
80	23.2	1.1	21.1	7.9	1.2	< 0.3	0.08	1.3	13.8	15.2	26.1	58	7.80	0.11
120	17.2	0.9	19.2	8.3	53.6	< 0.3	0.09	1.1	35.1	36.2	20.8	100	6.72	0.07
140	5.6	nf	nd	8.6	160.1	< 0.3	0.07	0.9	23.3	24.3	10.2	100	nd	nd
170	4.4	nf	nd	8.6	122.5	< 0.3	0.08	1.2	21.6	22.9	8.5	100	nd	nd

nf – not found nd – not determined

Figure 1. Soil carbon stores in the upper 0.5m of the studied soils being 8, 19, 37 and 59 years under self-restoration, of an actual arable land, and of a "Natural" Chernozem, never been cultivated

Table 2. Free particulate organic matter (POM) of the density fraction <1.8g/cm^3, occluded POM of the density fraction <1.8g/cm^3, and OC within the grain size fractions of the studied soils being 8, 19, 37 and 59 years under self-restoration, of an actual arable land, and of a "Natural" Chernozem, never been cultivated

	Sand	Silt			Clay		
Depth	0.2-0.063mm	0.063-0.020mm	0.020-0.0063mm	0.0063-0.002mm	<0.002mm	Free POM of the density fraction <1.8 g cm^{-3}	Occluded POM of the density fraction <1.8 g cm^{-3}
cm	% of soil						
Arable land							
10	0.003	0.02	0.04	0.17	1.40	0.12	1.69
20	0.005	0.02	0.04	0.17	1.85	0.07	1.26
40	0.005	0.01	0.02	0.15	1.30	0.03	0.88
50	0.002	0.01	0.01	0.10	1.18	0.03	0.68
8 years							
10	0.006	0.03	0.07	0.27	1.70	0.28	1.70
20	0.004	0.02	0.06	0.23	1.61	0.10	1.46
30	0.004	0.02	nf	nf	1.14	0.08	1.40
55	0.001	nf	0.01	0.07	0.99	nd	0.56
19 years							
10	0.005	0.03	0.05	0.19	1.37	0.11	2.19
20	0.007	0.04	0.07	0.2	1.62	0.04	1.18
30	0.010	0.07	0.04	0.17	1.69	0.02	0.94
50	0.020	0.07	0.06	0.15	1.36	0.01	0.71
37 years							
10	0.01	0.09	0.10	0.40	1.88	0.3	1.92
20	0.006	0.06	0.09	0.38	1.86	0.13	1.43
40	0.005	0.03	0.04	0.32	2.35	0.06	0.77
55	0.004	0.01	0.03	0.16	1.80	0.07	0.61
59 years							
10	0.006	0.13	0.07	0.28	1.84	0.57	2.65
20	0.004	0.07	0.07	0.24	2.16	0.10	1.90
50	0.002	0.02	0.05	0.25	2.26	0.07	1.03
"Natural" Chernozem							
10	0.02	0.09	0.08	0.63	2.42	0.6	4.05
20	0.005	0.06	0.09	0.33	2.20	0.35	1.85
60	0.002	0.03	0.03	0.16	1.46	0.15	0.95
80	0.004	0.02	0.02	0.06	0.95	0.05	0.75
120	0.007	0.07	0.08	0.15	0.74	0.04	0.51

The investigation in respect of functionally different SOC pools of post-agrogenic Chernozems under self-restoration reveal with 36-68% of SOC a dominance of organic carbon (OC) in the fine silt and clay fractions (Table 2). This fraction showed a tendency to increasing during self-restoration, hence indicating an increase of passive OC. An increase during self-restoration was also found in respect to POM, but the amounts were with 1-7% of SOC comparatively low. The fraction of the occluded POC was 20–58% of SOC. Although it also increased during self-restoration, the rate of occluded OC after 59 years of self-restoration didn't reach the level of the virgin Chernozem.

Conclusion

Self-restoration of post-agrogenic Chernozems in the Kursk steppe zone of Russia was characterized by reaching the climax stage for the vegetation after 59 years. Even quicker was the development towards Chernozem typical morphology, by loosing the ploughing features and gaining the crump structure, overall indicating a fast recovery from agriculture. But our results in respect of SOC and SOC fractions showed clearly, that the restoration process is not completed after 59 years: although a distinct increasing carbon sink functioning occurred, the question of when or ever the SOC dynamics of virgin Chernozems will be reached remains open. To fill this gap more investigations are required, which should also include other soils of other climate zones.

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Effect of charcoal (biochar) amendments in Manawatu sandy-loam soil (New Zealand) on white clover growth and nodulation

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Abstract
New Zealand primary industries such as dairying produce significant amounts of greenhouse gases, particularly methane. Storage of carbon in agricultural soils in the form of biochar has been proposed as a means of offsetting carbon emissions. Biochar could also improve soil conditions for plant growth. This paper reports the effects of biochar as a soil amendment added to a Manawatu sandy-loam soil, on the soil pH, soil carbon and nitrogen, mineralisable nitrogen and hot water extractable carbon, together with root and shoot growth for white clover (*Trifolium repens* var Emerald). Biochar was added to the soil in differing proportions. The proportions were 0:100, 20:80, 40:60, 60:40, 80:20 and 100:0, biochar to soil. Each of these six treatments was replicated four times and set up in a pot trial in a glass house configured with a Randomised Complete Block Design (RCBD). The addition of biochar reduced the growth of white clover shoot and root dry matter. The increased carbon content of soil and reduced supply of plant nutrients may be the reason for the reduced growth of white clover and biochar increased the pH.

Key Words
Biochar, carbon, soil properties, white clover, nitrogen, Randomised Complete Block Design (RCBD).

Introduction
The release of carbon (C) into the atmosphere far outweighs the fixation of carbon by soil organisms. Considering the incessantly increasing quantity of carbon dioxide (CO_2) in the atmosphere, retaining carbon in the soil is a concern now and in the future. A proposed simple way of retaining carbon in the soil is through addition of biochar. Biochar is a term reserved for the plant biomass derived materials contained within the black carbon (BC) continuum. This definition includes chars and charcoal, and excludes fossil fuel products or geogenic carbon (Lehmann *et al.* 2006). Materials forming the BC continuum are produced by partially combusting (charring) carbonaceous source materials, e.g. plant tissues (Schmidt and Noack 2000), and have both natural as well as anthropogenic sources. Depending on the temperatures reached during combustion and the species identity of the source material, a biochar's chemical and physical properties may vary (Keech *et al.* 2005). For example, coniferous biochars generated at lower temperatures, e.g. 350°C, can contain larger amounts of available nutrients, while having a smaller sorptive capacity for cations than biochars generated at higher temperatures, e.g. 800°C (Gundale and DeLuca 2006). However, the effects of biochar as a soil supplement are not fully understood; and need further investigation to determine how it confers benefits on soil.

Figure 5. White Clover in RCBD

A proposed way of testing biochar as a beneficial soil amendment involves investigating its effect on white clover growth. White clover (*Trifolium repens*; Figure.1) is a key component of New Zealand pastural agriculture, which is dependent on an inexpensive, high quality, feed source. In pastures, white clover provides a cheap, continual source of nitrogen (N), has high nutritive value, improves forage intake, utilisation rates of livestock, and complements perennial ryegrass growth. It is also considered environmentally friendly and therefore contributes to New Zealand's "clean-green" image. To add to this, white clover has a fast growth rate, making this species excellent to test benefits of biochar-supplemented soils in a limited time frame and in a pot study. The objective of this study was to measure the effect of charcoal (biochar) amendments in Manawatu sandy-loam soil (New Zealand) on white clover growth and nodulation and also changes to soil total carbon and nitrogen, pH, Hot water extractable carbon (HWC) and mineral-N (nitrogen) content.

Methods

Production of the charcoal powder (biochar)

The source material is Malaysian Mangrove wood as a renewable resource, *Rhizophora* species, preferably *Rhizophora apiculata*. It is a common tree found in swamps, especially at river mouths. Mostly only thirty year old mangroves are harvested for logs. The Charcoal that is made from mangrove wood has a strong and high density structure; the charcoal is hard and heavy. The logs are kilned at a temperature of 220°C. The first stage of the kilning process takes around 8 to 10 days. The log condition inside the kiln is determined by the smoke that comes out of the holes of the kiln. After 10 days the kiln is completely shut off and the baking process continues on a temperature of around 83°C. This takes another 12 to 14 days. Then the cooling process starts, this takes another 8 days before the hole in the kiln is opened. The material is then crushed to various sizes and powders

Soil and Plant measurements

Manawatu sandy-loam soil of the Recent Soil group ('Dystric Fluventic Eutrudept' in US Soil Taxonomic Classification as reported by (Hewitt 1998) was collected and mixed in appropriate proportions with the biochar by weight (Table 1). The mixtures were sieved, and left for one month before sowing. The white clover seeds (0.1 g) were sown into each of the soil mixes on 2nd October 2009 and regularly watered to maintain soil moisture. The white clover foliage was harvested after 93 days of growth (13th January 2010) and after 111 days of growth (31st January 2010) the whole plant was removed from the treatment medium, separated into shoots and roots and oven-dried at 70ºC for 72 h. A sample of fresh root was fixed in FAA (contains ethanol (70%), formaldehyde and acetic acid at a ratio of 90:5:5 by volume) to count nodulation.

Table 2. Project Treatments

Treatment	Biochar:Soil ratio
B0S5	0:5
B1S4	1:4
B2S3	2:3
B3S2	3:2
B4S1	4:1
B5S0	5:0

Soil total Carbon and Nitrogen, Hot water extractable carbon (HWC) (Ghani *et al*. 2003), mineral-N (nitrogen) content (Keeney and Nelson 1982) and pH in water were measured at the start and at the end of the experiment.

Results

Shoot and root dry matter biomass

In the first harvest, plant dry matter harvested in treatment B0S5 was significantly higher than treatments B1S4, B2S3, B3S2 and B5S0 but treatments B2S3, B3S2, and B4S1 had the same shoot growth. Treatment B5S0 inhibited growth of white clover but did not kill the plants (Figure 2). At the final harvest also B0S5 showed significantly higher shoot growth than treatments B1S4, B2S3, B3S2, B4S1 and B5S0. When compared to the treatment B1S4, B2S3, B3S2 and B4S1 showed higher shoot growth (Figure 3). The roots also showed similar trend in treatments B0S5, B2S3, B3S2 and B4S1 like shoot growth at final harvest (Figure 4) but B1S4 did not but was similar to B2S3.The final dry mass of shoot and root suggested that the addition of biochar reduced the yield of white clover. However, there is a trend in increment of dry matter on root from B1S4 to B3S2 suggesting better combination effect on growth and detailed study required to confirm this effect.

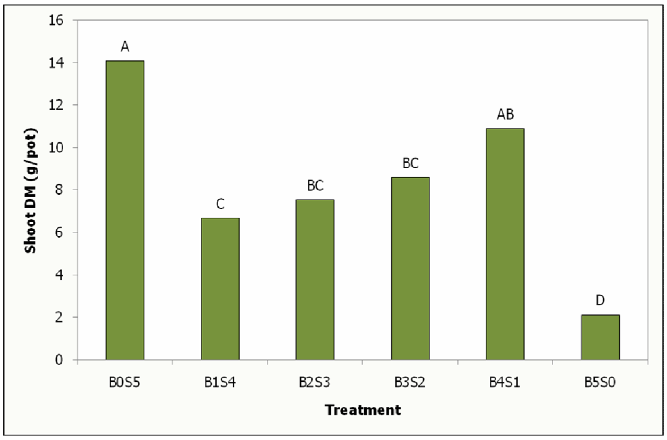

Figure 6. Clover shoot dry matter (DM) at first harvest at different treatment levels (Bars with different letters are significantly different (P<0.0001))

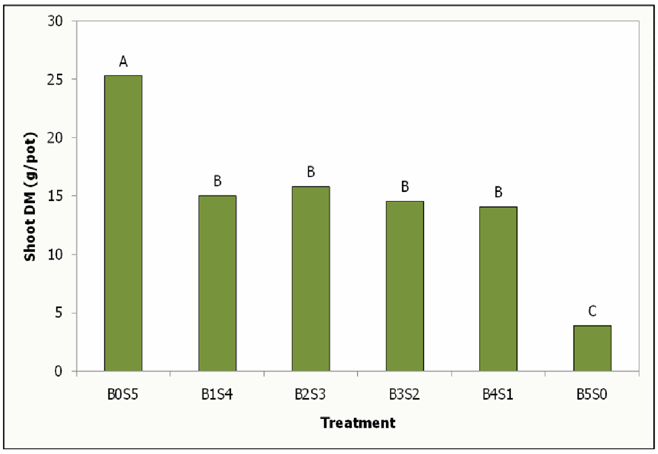

Figure 7. Clover shoot dry matter (DM) at final harvest at different treatment levels (Bars with different letters are significantly different (P<0.0001))

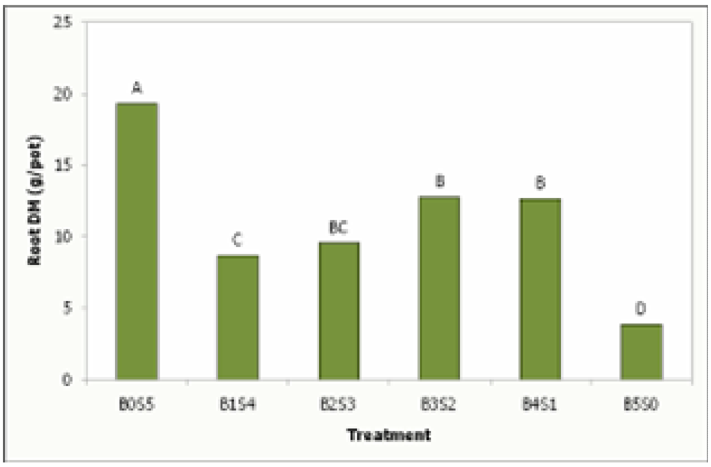

Figure 8. Clover root dry matter (DM) at different treatment levels (Bars with different letters are significantly different (P<0.0001))

Soil properties

Total carbon

The percentage of soil carbon increased from 1.14 to 40.81 as the addition of biochar to soil ratio was increased. The increased carbon content of soil may be the reason for the reduced growth of white clover (Table 2) or the reduced supply of plant nutrients (e.g. N) resulting from the reduction in soil.

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Table 2. Measurement of total carbon, pH and Nitrogen mineralisation

Label	% of soil carbon		Initial pH		Final pH		Initial Nitrogen (N) mineralisation (µg/g)		Final Nitrogen mineralisation(µg/g)	
B0S5	1.14	(±0.01)	5.49	(±0.05)	5.16	(±0.05)	129.67	(±9.42)	2.90	(±0.40)
B1S4	11.5	(±0.36)	7.04	(±0.03)	7.26	(±0.03	102.13	(±3.12)	6.49	(±1.02)
B2S3	24.43	(±3.03)	6.47	(±0.07)	7.28	(±0.06)	78.03	(±22.95)	6.77	(±1.41)
B3S2	31.28	(±0.80)	6.06	(±0.05)	6.88	(±0.07)	132.10	(±24.99)	7.00	(±1.13)
B4S1	35.79	(±1.03)	6.09	(±0.02)	6.81	(±0.04)	85.53	(±14.69)	9.29	(±1.68)
B5S0	40.81	(±0.42)	6.40	(±0.12)	7.64	(±0.05)	62.74	(±15.76)	10.05	(±0.83)

pH in water

The initial pH on addition of biochar varied from 5.49 to 6.4 and the final pH was from 5.16 to 7.64, confirming that the addition of biochar has increased the pH (Table 2).

Conclusion

In this investigation the addition of biochar to soil reduced the biomass of both white clover shoot and root. The increased carbon content of soil and reduced supply of plant nutrients may be the reason for the reduced growth of white clover and biochar has increased the pH. Further research is needed in the area of addition of biochar to different soil types.

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Effect of soil management and crop rotation on physical properties in a long term experiment in Southern Brazil

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Abstract
No-tillage system associated with crop rotation increases the amount of crop residues left as mulch on the topsoil, and can be an important and sustainable alternative for soil management in tropical and subtropical conditions. The objective of this work was to evaluate the soil physical properties affected by cover crop, rotation and soil management in a long-term experiment in South Brazil. The experimental site was cultivated for 10 years in a conventional system. Subsequently, the field experiment was established in 1986, and treatments combined six winter cover crop species, and two tillage systems (conventional tillage - CT and no-tillage - NT). The treatments were laid out using a split-plot design in three blocks. The soil samples were collected in October 2005 from trenches at six depths. The bulk density, aggregate size fractions and porosity were determined. The NT system contributed to increase the formation of coarse aggregates and improved the soil aggregation parameters. In the first upper layers, the soil disturbance due to ploughing every season, enhanced the macroporosity and diminished the microporosity on conventional system comparatively to NT. Independently of the type of soil management all winter species increased the higher aggregate size class.

Key Words
Conservation tillage, subtropical soil, sustainability, soil organic matter, cover crop.

Introduction
Soil organic matter (SOM) plays a key role in the formation and stabilization of soil aggregates (Oades 1984). Nevertheless, it was observed changes in aggregate stability following land use changes without changes in total SOM content (Puget *et al.*1999). Normally the macroaggregates are more susceptible to physical disruption because of the labile nature of binding agents (Hussain *et al*. 1999). Persistent binding agents are important in microaggregation (< 250 µm) of soil (Tisdall and Oades 1982). In the no-tillage system (NT) introducing winter crops into the rotation can enhance the soil aggregate size and stability when compared to winter fallow treatment (Calegari and Pavan 1995). Castro Filho *et al.* (2002), working during 21 years in a clay distrophic red latosol in north Paraná Brazil, found the best aggregation indices for the 0-20 cm layer in the NT system, mainly due to the increase in the organic carbon content. Biological activity, enhanced by NT (Holland 2004) plays a crucial role in aggregate stabilization. The root exudates can produce cementing agents which can strongly adsorbed to inorganic materials, thereby helping to stabilize soil aggregates (Tisdall and Oades 1982). Further, both accumulated carbon and root exudates increase soil microorganisms activity with subsequent production of microbial binding agents. Also the roots physically can influence aggregation by exerting lateral pressures inducing compaction, and by continually removing water during plant transpiration, leading to localized drying of the soil and cohesion of soil particles around the roots (Six *et al*. 1999). These effects in the soil are more important than the shoot residues left on soil surface in the formation of aggregates and stabilization of aggregate-associated SOM (Puget *et al.* 1995; Gale *et al*. 2000).

The main objective of this work is to evaluate such soil physical properties affected by cover crop rotations and soil managements NT and CT systems, after 19 years cultivating in a clayey Oxisol in South Brazil with different winter cover crop species.

Methods
The field experiment was established in 1986 in the IAPAR (Agronomic Institute) Experimental Station at Pato Branco, southwestern Paraná State, Brazil (26°7'S, 52°41'W, and 700 m altitude). Climatologically the area belongs to the sub humid tropical zone (Köeppen's Cfb) with a climate without dry season, fresh

summer and an average of hottest month lower than 22 °C. Annual rainfall average ranges from 1200 to 1500 millimeters per year. The soil of the experimental site is a clayed Oxisol acid with a high clay content (72 percent clay, 14 percent silt, and 14 percent sand), inherited from basaltic materials and thus containing kaolinite and iron oxides minerals (Costa 1996).

The experimental site was before cultivated for 10 years in a conventional system mainly with maize and bean. Experimental treatments combined six winter species, and two tillage systems, i.e. conventional tillage and no-tillage, in order to obtain a large variety of situations in term of biomass inputs and soil management. The conventional tillage consists in one disc plough and two discs harrowing two times a year before summer and winter crop planting. The winter specie treatments were blue lupin (*Lupinus angustifolius* L.), hairy vetch (*Vicia villosa* Roth*)*, black oat (*Avena strigosa* Schreb), oilseed radish (*Raphanus sativus* L.), winter wheat (*Triticum aestivum* L.), and fallow. Except wheat the other cover crops were controlled at the flowering stage cutting with a knife roller (lupin, hairy vetch, black oats and oilseed radish) or by the application of glyphosate (fallow) some years after the knife-roller it was complemented with herbicide. After the wheat grain was harvested a mat of dead material was left on the surface of the soil as mulch or incorporated before planting the summer crop. From the winter 1986 until October 2005, the biomass production (winter species) was evaluated at the flowering stage (management time) and also the amount of summer crop residues left on the soil surface after harvesting. Summer crops of maize and soybean were planted during each year following the winter species.

Lime prepared with dolomite was applied 5 times on a total of 9.5 Mg/ha (1.0, 2.0, 3.0, 1.5 and 2.0 Mg/ha of lime in all plots, in 1989, 1992, 1995, 1999 and 2001, respectively). On the NT system the lime was broadcast on the soil surface and in the CT was incorporated by ploughing. The summer crops received chemical fertilizer every year, and the total amount of chemical fertilizer applied during 19 years were, in kg/ha: 1300 of P_2O_5, 745 of K_2O, and 425 of N for corn only. Generally the fertilizer was applied at planting time (P and K), and for N 1/3 at planting time and 2/3 at 45 days after planting of corn.

The soil samples were collected in October 2005 from trenches at six depths: 0-5, 5-10, 10-20, 20-30, 30-40, and 40-60 cm. Bulk density measurements were made on steel cylinders (5 cm diameter). Soil texture was measured by the pipette method. The soil sample was dried in the oven at 105 ºC to determine water content. The mean weight diameter (MWD), geometric mean diameter (GMD), aggregate stability index (AS%) of soil aggregate, microporosity (MIP, pores < ~50 µm), soil density, and particle density were measured. The total porosity (TP) and macroporosity (MAP) was estimated.

The treatments were laid out using a split-plot design in three blocks. The winter species were the main plots (240 m^2) and the tillage treatments were subplots (120 m^2). Also soil samples were taken in a forest on the border of the experiment just for comparing. The experiment statistical analysis considered the winter crops as a main plot, the soil management (NT and CT) are the subplot, and the soil layers the subsubplot (trifactorial) (2 soil management x 6 cropping sequence x 5 or 2 soil layers). The mean values were compared by the Test of Least Significant Differences (LSD) when the analysis of variance was significant.

Results

Plant Residues

The total amount of organic residues (winter crop, maize and soybean residues) added to the soil during the 19 years of this experimental study has been reported in Calegari *et al*. (2008). The total amount of winter crop residues was higher for no-tillage treatment compared to that of conventional tillage. Because the fallow treatment is not cultivated during the winter season, it produced fewer residues than other winter treatments in both NT and CT systems. Thus, the winter crop is the main factor that contributed to differentiate the amount of biomass introduced to the soils.

Soil bulk density

The soil bulk density values for forest were lower than 1.01 kg/dm^3 in the all soil layer sampled (Table 1). The bulk density at arable layer (0-30 cm) increased to, in average, 1.2 kg/dm^3. In the deeper soil layers assessed, the bulk density remained very close to the natural condition, even if the soil has been disturbed intensively (38 times by plough and 76 times by disc harrowing) during 19 years of cultivation. Bulk density of the soil layers is very similar between CT and NT treatments, except in 10-20 cm layer where bulk density is a little higher under conventional tillage. No significant difference can be observed between winter crop treatments.

Soil particle aggregate size distribution

In the NT system the proportion of big aggregates (> 2 mm) is higher than in CT, for both topsoil layers (0-10, 10-20 cm), and the reverse for other fractions (Table 2). Thus, the NT system is characterized with the formation of coarse aggregates and the CT with finer aggregates. There is a gradient between 0-10 and 10-20 cm in most of the aggregate size under no till, while this difference was not observed in CT. Winter crop treatments show a lower proportion of big aggregates (>2 mm) and a higher proportion of small aggregates (<0.25 mm) in fallow compared to other treatments. Thus, the winter fallow treatment, which had the lowest amount of organic residues added during the all 19 years (Calegari *et al*. 2008), is characterized by finer aggregates. No significant differences were found between winter crops.

Probably in our site, the effects of winter crop root systems, and the lack of soil disturbance during 19 years in NT contributed to promote higher soil aggregation classes (> 2.00 mm) than fallow and CT. From 13 to 24% of the soil organic carbon was considered to be physically protected against biodegradation due to its location in clay or silt sized microaggregates.

Table 1. Bulk density in clayed Oxisol of Brazil affected by soil management and crops system.

Soil management	Soil layer (cm)				
	0-10	10-20	20-30	30-40	40-60
	(kg/dm^3)				
NT	1.25aA	1.19bB	1.11bC	1.06aD	1.00aE
CT	1.24aB	1.28aA	1.17aC	1.08aD	1.02aE
* Forest	0.80	0.90	1.00	1.00	1.00

Means followed by the same lower case letters in the same column comparing soil management for each depth into the same aggregate size class , and also winter treatment into each aggregate size class, do not differ at the 5% level of probability by the *F*-test in the analysis of variance. Means followed by the same upper case letters in the same line comparing depth into each soil management into the same aggregate size class do not differ at the 5% level of probability by the *F*-test in the analysis of variance.
* Forest soil is not included in statistical analysis

Table 2. Aggregate size class distributions in clayed Oxisol in Brazil affected by soil management and crops system.

Soil layer (cm)	Soil management	Aggregate size (mm)				
		> 2.00	2.00-1.00	1.00-0.50	0.50-0.25	< 0.25
		(g/100g)				
0-10	No-tillage	39.97 aA	6.65 bB	5.28 bB	3.45 bB	4.92 bB
	Conventional	22.39 bA	12.30 aA	11.38 aA	7.23 aA	9.29 aA
	* Forest	28.37	8.78	8.93	5.24	5.81
10-20	No-tillage	36.08 aB	8.40 bA	7.37 bA	4.34 bA	4.85 bB
	Conventional	23.76 bA	11.73 aA	11.57 aA	6.59 aB	8.63 aA
	* Forest	31.05	9.05	7.77	5.47	6.07
Winter treatment						
	Fallow	27.31 b	10.18ns	9.36ns	5.82ns	8.91 a
	Vetch	31.83 a	9.88	9.03	5.38	6.08 b
	Lupin	31.31 a	9.58	8.59	5.18	6.47 b
	Radish	31.26 a	9.21	8.76	5.27	6.92 b
	Oat	31.14 a	9.98	8.74	5.23	6.41 b
	Wheat	30.45 a	9.80	8.93	5.53	6.74 b

Means followed by the same lower case letters in the same column comparing soil management for each depth into the same aggregate size class , and also winter treatment into each aggregate size class, do not differ at the 5% level of probability by the *F*-test in the analysis of variance. Means followed by the same upper case letters in the same column comparing soil layer into each soil management into the same aggregate size class do not differ at the 5% level of probability by the *F*-test in the analysis of variance.
* Forest soil is not included in statistical analysis

Soil aggregation parameters

In the two soil depths studied (0-10 and 10-20 cm), the three soil aggregation parameters (MWD, GMD and AS) enhanced under the NT system. In this system the values found for MWD and GMD were superior in the upper layer (0-10 cm) when is compared with the beneath soil layer (10-20 cm) whereas, the MWD increased with depth in CT system where the soil and organic residues are incorporated into the soil. Values of MWD, GMD and AS were found lower in fallow compared to winter crops but no significant differences were found between winter crop species. The relative enhance promoted by NT compared with CT was of 78% for MWD, and 238% for GMD. The comparison at beneath soil layer (10-20 cm) also presented the same trend observed previously in upper layer, with 51%, for MWD, and 83%, for GMD, favorable to no-tillage. For the AS %, independent of soil depth, presented 7% higher under NT than under conventional.

Soil porosity

In the first upper layers (0-10 and 10-20 cm), the soil disturbance by plough every season, enhanced the macroporosity on conventional system compared to no-till. But at beneath layers from 20 cm to 60 cm soil depth, where there are no more effects of plough, NT presented higher macroporosity compared to CT. These differences are significant at 0-10, 20-30 and 40-60 cm soil depth. The higher values for microporosity were found on NT at the upper layers (0-10 and 10-20 cm), but below this no difference were found among the two soil management systems. The comparison among the different winter crop treatments showed no significant differences of macroporosity (despite a lower value in the fallow treatment) and a significantly higher microporosity on the oat and the fallow treatments. As a result, total porosity in NT system was lower at 0-10 cm soil depth, higher at 10-30 cm and similar at 30-60 cm, compared to CT system. Total porosity is often reduced under NT compared with CT in comparison of < 10 years; but conversely in our experiment with 19 years NT presented higher total porosity.

Conclusions

The high amount of crop residues added to the soil during the years improved the soil aggregations parameters, and NT not promoted soil compaction, and the fallow treatment presented the lowest values for MWD, GMD and also for AS%. Furthermore, under conventional system, the soil disturbance by plough every season, enhanced the macroporosity and diminished the microporosity on conventional system comparatively to no-tillage, and promoted the formation of smaller diameter classes.

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Estimating the carbon sequestration potential of agricultural soil reforested with directly seeded native vegetation belts around Canberra, Southern Tablelands, NSW

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Abstract

Sowing native vegetation tree belts on agricultural soils using direct seeding is a technique that has been carried out by Greening Australia in the Australian Capital region for many years. It results in improved landscape biodiversity and habitat, provides shelter for pastures, crops and livestock and protects soil from erosion. Initial research identified a range of changes the vegetation belts make to the physical and chemical functions of soils including: an increase in the thickness of the surface litter and depth of the A_o horizon. This leads to a commensurate increase in total soil organic C; a reduction in bulk density; an increase in total nitrogen; and a decrease in pH and effective CEC. The study also used the Landscape Function Analysis method and found a significant improvement in the three functional indices: infiltration, nutrient cycling and stability. These findings are being investigated as part of a new study which aims to develop a model to estimate the C sequestration potential of agricultural soils in which vegetation belts have been established. Additionally, the chemical nature of the soil organic C in the vegetation belts will be compared with C from adjacent grassland soils to identify whether the trees are sequestering a more stable form of soil C than the grasslands. The soil biological and ecological characteristics are being analysed as part of the research.

Key Words

Soil organic carbon, SOC, reforestation, carbon sequestration

Introduction

Since the arrival of Europeans in Australia the landscape has undergone considerable change due to widespread clearing of native vegetation ecosystems to make way for agricultural activities. The removal of the native vegetation has contributed to substantial soil degradation and erosion across the landscape. In an effort to restore some landscape ecosystems, Greening Australian has established a number of native vegetation tree belts on agricultural land across the Southern Tablelands of New South Wales. The belts have a range of ecological benefits, including enhancement of landscape biodiversity and habitat and provision of shelter for pastures, crops and livestock. They can be a source of timber or fodder but importantly they protect soil from the erosive forces of wind and water and improve soil ecological function.

In 2008 an initial study was carried out on 36 native vegetation tree belts sown by Greening Australia using a direct seeding method during the period 1990 to 1996. The study identified changes to soil physical and chemical functions attributable to the presence of vegetation belts. This research was undertaken by comparing a range of soil function variables in the native vegetation belts with those on soil in the adjacent grassland/paddock. The grasslands were presumed to represent the condition of the soil prior to the sowing of the belts and consequently to give an indication of the extent and direction of the soil changes initiated by the belts. For each vegetation belt an analogue site was also selected to be a benchmark of the soil condition. The analogue sites were presumed to be the best representation available of the original soil condition in the grasslands prior to intervention by land clearing and subsequent agricultural activity, and consisted of laneways, travelling stock reserves or road verges.

Methods

A range of soil physical and chemical attributes were analysed. 36 sites were measured in Phase 1, and then a subset of six were studied in greater detail in Phase 2 (Table 1).

Table 1. Soil attributes studied during in 2008 during Phase 1 and Phase 2.

Attribute	Phase 1 (36 sites)	Phase 2 (6 sites)
Soil Landscape	✓	
Soil Profile (Northcote)		✓
Surface litter (t/ha)	✓	
A_O	✓	
SOC (Walkley & Black) for A1 horizon (phase 1) and B horizon (phase 2)	✓	✓
Bulk Density		✓
C Density (to 30cm)		✓
Biota abundance		✓
pH / EC (1:5 water)	✓	✓
Total N & P		✓
Extractable Anions		✓
Total Cations		✓
Exchangeable Cations		✓
Effective Cation Exchange Capacity		✓
Landscape function analysis	✓	

In Phase 1, 10 soil samples from the A_1 horizon to a depth of 5cm were collected from each of the 36 sites with 2 samples obtained from the grassland above and 2 from below the vegetation belt, 4 from within the vegetation belt (two from the tree-row and two from the inter-row), and two from the analogue site. The landscape function analysis (LFA) method (Ludwig and Tongway, 1995) was also used to record changes to landscape functionality.

Five samples of each of the A_1 and B horizons were obtained from the six phase 2 sites with one sample obtained from the grassland above and below the vegetation belt, 2 from within the vegetation belts (one from the tree-row and one from the inter-row) and one from the analogue site.

Results

There were 20 soil-landscape units indentified across the 36 sites. This variation may have influenced the distribution of the soil function results between sites, but no relationship could be identified, probably due to the low total number of study sites compared with the diverse range of soil landscape units. The research findings from Phase 1 indicated that compared to the adjoining grasslands, the tree belts had significantly ($P<.001$) greater amounts of surface litter, a more developed A_o horizon and a higher percent of total organic C.

The LFA results demonstrated an improved infiltration, stability and nutrient cycling capability for the soils within the vegetation belts when compared with the adjacent grassland/paddock and that the infiltration capacity within the vegetation belts had exceeded that of the analogue sites. Landscape Function Analysis also indicated an improved soil function down slope of the vegetation belts compared with upslope. Sites in which some livestock grazing had occurred within the tree belt had a higher TOC value indicating a beneficial outcome in terms of nutrient cycling. It is probable that excreta from stock and trampling breaks down the surface litter and thereby hastens the decomposition process.
Phase 2 results indicated that there were no significant differences in total N and total P between the belts and adjacent grassland/paddock. However, N values tended to be higher within the vegetation belts; this is thought to be caused by the symbiotic fixation of N due to the presence of the native leguminous *Acacia* spp. trees in the vegetation belts, and P values were lower in the belts, probably due to the extraction of P by the trees. The pH was marginally lower within the vegetation belts, but this trend was not significant.

Discussion

Agricultural soils have the capacity to lose or gain C depending upon management practices used. For example, when organic matter is removed from the system such as when land is converted from forest to agriculture there can be a reduction in SOC by up to 50% (Lal 2005). However, improved land management practices such as reduced tillage have been shown to increase SOC (Six *et al.* 1998). Inputs of organic matter from plant material contribute to an increase in C stock in soils, as does an increase in clay content, although climatic conditions such as moisture and temperature can influence the rate of decomposition and subsequent loss of SOC (Jastrow *et al.* 2007). Forest soils gain a large proportion of their organic matter and SOC from tree litter; therefore it is beneficial to soil function to establish trees in agricultural soil to increase the SOC stock (Read 2008), although other studies have found that in some cases this can take many years depending on the level of soil disturbance during establishment (Turner 2005).

Both native and plantation forests and including those used for agroforestry store C in both the above ground biomass and contribute C to the SOC pool. For C accounting purposes, reforestation is determined by the Australian Government (CPRS White Paper 2009) as meeting the Kyoto Protocol when the trees have been established by humans on land that was clear of forest on 31 / 12 / 1989. The trees must be determined to have the potential to grow to a height of at least two metres, have a crown cover of at least 20% and be grown on areas larger than 0.2ha. The vegetation belts sown by Greening Australia, fulfil the definition of reforestation under the terms of the Kyoto Protocol and potentially are an important contributor to the SOC pool.

Measuring above ground living biomass is commonly used as a surrogate for calculating the C content in forest communities. Each ecosystem has a unique biomass concentration and therefore measurements of the above-ground biomass which can include both overstorey and understorey biomass and standing woody debris can be used for estimating C in forest ecosystems, and fallen debris, surface litter, root systems and SOC are measured to determine forest SOC stocks (Snowdon *et al.* 2002).

In this follow-up study, measurements of the above ground biomass of direct seeded native vegetation belts will be correlated with soil classification and SOC density results and will be used to develop a model for predicting the potential of certain soil types as found across the Southern Tablelands of NSW to sequester C. Organic matter has many roles (e.g., water retention, physical, chemical and biological properties of soil), but it is also integral to the process of developing and stabilising soil aggregates. The stabilisation process involves protecting SOM from mineralisation by microorganisms. Physical protection of SOM is achieved by encapsulating SOM within aggregates, and micro aggregates (<250 µm) provide greater protection than (macro aggregates >250 µm) (Goebel *et al.* 2009).

Macro aggregates are more sensitive to cultivation and land use change than micro aggregates but can be stabilised by the presence of carbohydrate rich root or plant debris being occluded within the aggregates (Six *et al.* 1998). The LFA results from the preliminary study show that for the stability index soil from the vegetation belt has improved compared with that analysed from the grassland soil. To determine whether there is a link between the improved soil stability index and aggregate stability the physical nature of aggregates in soils in the vegetation belts will be compared with those from the grasslands to determine whether sufficient time has elapsed to allow micro aggregate development and if so also determine the extent the aggregates are contributing stable C to the soil pool.

Polysaccharides such as cellulose, hemi-cellulose, chitin and peptidoglycan can be found in fresh plant and microbial tissues in soil particles > 20µm. Decomposition of the labile components result in a reduction in particle size and an increase of more recalcitrant materials such as lignin and alkyl structures from the 2-20µm size (Baldock *et al.* 2000). The follow-up study will analyse the chemical structure of the SOC from both the vegetation belts and grasslands to determine whether the nature of the SOC is changing and becoming more stable.

Conclusion

Overall, the results of the preliminary research project provide evidence that the vegetation belts can induce an improvement to a range of soil function variables. However, over time as the tree growth declines there is evidence that surface litter and TOC also may decline. These trends are being investigated in a larger 3 year follow-up study where the changes to SOC are being examined more closely to investigate whether it is possible to predict the amount of increase in SOC that can be attributed to the establishment of the tree belts and whether soil type or tree species mix has any influence on the extent of change. This will be done by comparing tree biomass with SOC density to 30cm depth at approximately 100 sites across the Southern Tablelands of NSW, and then comparing these results with biomass and SOC density results from adjoining grasslands.

Other important objectives of the follow-up study are to determine whether there has been sufficient time since the establishment of the vegetation belts for aggregates to stabilise and provide greater physical protection for the organic matter within the vegetation belts compared with the grasslands and whether the chemical nature of the SOC is changing to increase the concentration of stable C.

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Farming soil carbon calculator (FSCC) – Estimation of soil carbon by improved land management practices in Central West NSW

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Abstract

Excel spreadsheet 'Farming Soil Carbon Calculator' (FSCC) has been developed to determine the expected soil carbon levels for different soil types, land management practices and climatic zones throughout the Central West Region of NSW. This simple tool was developed to rank farmers, so that those increasing soil carbon levels could receive a financial incentive to continue their improved land management practices. This poster paper explains the logic that drives the spreadsheet, providing three examples of a grazier on the tablelands, a mixed cropping and grazing enterprise on the slopes, and a cropper on the plains.

Key Words

Farming, soil carbon, calculator, spreadsheet, land management, incentive payment, Central West NSW.

Introduction

Since 2004, the Central West Catchment Management Authority (CMA) has implemented hundreds of on-ground projects in conservation farming machinery, perennial pasture establishment, improved grazing management through fencing and water projects and the application of sodic soil ameliorants. A total of 582,777 ha farming lands throughout the Central West catchment have been managed for improved soil health (CWCMA 2008). By working closely with relevant government agencies and regional organisations such as Carbon Farmers of Australia, the Central West CMA has promoted the benefits of 'carbon farming'. In the past three years the Central West CMA has conducted 'Carbon Cocky' competitions and sponsored three conferences to create awareness and flag the possibility of a voluntary 'Carbon Credits' scheme. Carbon credit schemes for agricultural lands is problematic because it is costly to accurately measure and monitoring soil carbon on farms, and there is a need to account for the green house gas emissions.
In 2009-10, the Central West CMA commenced a project titled 'Land Management Activities for Increasing Soil Carbon' to provide financial incentives for landholders who are practicing improved land management activities that increase soil carbon on their properties (Lawrie *et al.* 2010). To facilitate this project, an excel spreadsheet FSCC was developed to calculate the amount of soil carbon sequestered under different farming systems, soil types and climate zones throughout the Central West Region of NSW. The Central West CMAs Increase Soil Carbon project and FSCC spreadsheet is an initial step in developing a practical and affordable methodology that rewards farmers for the soil carbon sequestered. The main advantage of this approach is that it allows the limited CMA incentive dollars to be allocated to eligible farmers in an 'open and transparent' manner, and can be easily modified and refined as further soil carbon farming data becomes available.

Methods

The spreadsheet is based on the maximum known soil carbon levels for various soil types found throughout the region (Murphy *et al.* 2003) which is then linked to the 'best soil management practices' that can be implemented on agricultural lands. The formulas used to estimate the Soil C levels for the different scenarios, are based on the following logic:

- Clayey soils on the Slopes can store up to 70 Mg Soil C /ha under Woodlands, 50 tonnes / ha under Pastures and 35 Mg/ha under Cropping. Soil C on the Tablelands is multiplied by 1.3 to reflect improved soil sequestration under the cooler / wetter climate, while values for the Plains are multiplied by 0.65 reflecting the drier / hotter conditions.
- Soil clay content has a big bearing on the amount of Soil C that can be sequestered. Fine Textured Soils have a factor of 1 as these soils have the greatest potential to store soil carbon, Medium Textured Soils have a factor of 0.85, Coarse Textured Soils have a factor of 0.70 and Sodic Surface Soils have a factor of 0.60.
- Total Soil C (Mg /ha) and Soil C % calculations are based on 0 - 30cm depth which is the Kyoto protocol depth, with an assumed bulk density of 1.3.
- The spreadsheet takes into account the various management practices (Geeves *et al.* 1995; Charman

et al. 2000) that affect soil carbon levels under Pasture Land, Cropping Land and Woodland Areas which are described below. Each of the Land Management Actions has a range from 'Optimum Management' to 'Poor Land Management' and only one option can be selected for each action.

Pasture Lands

1. Grazing Management

a. Optimum Time Control Grazing for environmental outcomes (80 to 200+ days) with grazing charts
b. Time Control Grazing for production / livestock outcomes (80 to 200+ days) with grazing charts; Remove stock less than optimal circumstances
c. Optimal Rotational Grazing (20 to 80 days rest) Ground cover 70% - 80%; Pasture Biomass 1.0 Mg /ha; Pasture height 10cm - 15cm
d. Rotational Grazing (20 to 80 days rest); Remove stock less than optimal circumstances Ground Cover substantially < 70%
e. Optimal Set Stocking Stock numbers adjusted to pasture production
f. Set Stocking Stock numbers not adjusted to pasture production (overstocking)

2. Soil Nutrition

a. Optimal Management of Soil Nutrition Factors include Grazing Legume/Grass mixes, Regular Soil Testing, Lime, Sulphur and Phosphorus applications
b. Minimal Management of Soil Nutrition Factors include Legume/Grass mixes, Occasional Soil Testing, Some Lime, Sulphur and Phosphorus applications when affordable
c. Soils not Management for Soil Nutrition Minimal / No Soil Testing, No Lime, Sulphur and Phosphorus applications (obvious nutrient deficiencies / acidity), use of volunteer pastures

3. Soil Structure Management

a. Optimal Management of Soil Structure Use Ameliorants (Lime, Gypsum, Bio-solids), Regular Soil Testing, Managing stock to prevent compaction e.g. restricted grazing in wet areas
b. Soil structure problem not being addressed effectively

4. Pasture Species Composition

a. Well established perennial grasses Approx. 50% perennially and approx. 30% legumes
b. Pastures dominated by annual species < 50% perennially, and little or no legumes

5. Extensive Tree & Shrub Establishment

a. Saltbush Plantings / Alley Farming Systems / Agroforestry
b. No extensive tree and shrub establishment

6. Catchment Engineering Works

a. WaterPonding / Water Spreading / Natural Sequence Farming / Keyline / Extensive Soil Conservation Structures
b. No catchment engineering works

Cropping Lands

1. Tillage Practices

a. Zero Till Cropping (disc) < 5% topsoil disturbance
b. No Tillage 5% - 20% topsoil disturbance Narrow knife tynes or disc at sowing
c. Direct Drill One pass full disturbance at seeding
d. Reduced Tillage One pass before seeding, and full disturbance at seeding i.e. two cultivations
e. Multiple Tillage Two or more cultivation before seeding, and full disturbance at seeding

2. Stubble Management

a. 100% Stubble Retention (no grazing)
b. 50% to 80% Stubble Retention (grazing management, Coolamon harrows etc.)
c. 30% - 50% Stubble Retention (late, cool burning, baling)
d. Complete Stubble Removal (early hot burn)

3. Soil Nutrition

a. Optimal Management of Soil Nutrition Factors include Legume Rotation, Regular Soil Testing / Applications of N, P, K, S and Lime / Gypsum for Maximum Yield
b. Minimal Management of Soil Nutrition Occasional Soil Testing, Some N, P, Lime, Sulphur applications and some Legumes in rotation
c. Soils not Management for Soil Nutrition Minimal / No Soil Testing, No Nitrogen, Phosphorus, Lime, Sulphur applications (obvious nutrient deficiencies / soil acidity)

4. Soil Structure Management

a. Optimal Management of Soil Structure Use of Ameliorants (Lime, Gypsum, Bio-solids), Regular Soil Testing
b. Soil structure problem not being addressed effectively
c. Pasture / Crop Rotation
d. Pasture Cropping Direct seeding annual crop into a perennial pasture
e. Mostly well established perennial pastures > 50% time e.g. 6 years pasture, 4 years cropping
f. Mostly annual cropping or volunteer species > 50% time cropping e.g. 4 years pasture, 6 years cropping
g. Continuous Cropping

5. Cover Cropping

a. Cover Cropping Planting cover crop straight after harvest, then spraying / rolling before next years sowing
b. No Cover Cropping

6. Double Cropping

a. Double Cropping Sowing second crop immediately after harvest (opportunity cropping when seasonal rainfall is favourable)
b. No Double Cropping

7. GPS Controlled Traffic

a. GPS Controlled Traffic Machinery used confined to same wheel track
b. No GPS Controlled Traffic

Woodland Areas

a. Woodland Protection Livestock Excluded, Zero Cropping / Grazing, Ground Cover > 90%, All structural layers present (Trees, Shrubs and Grasses)
b. Woodland Management Strategic Grazing, Ground Cover > 70%, Area managed for production / biodiversity outcomes
c. Woodland Management Grazing, Ground Cover < 70%, Area managed for production

Each of the above land management practices were weighted by an expert panel within the Central West CMA to reflect their contribution to sequester soil carbon by:

- Increasing biomass production for improved groundcover and a biological food source
- Reducing soil disturbance and compaction
- Balancing soil chemistry and nutrition for optimum plant growth
- Increasing pasture or crop perenniality and / or increasing the rooting depth of annual plants such as crops
- Increasing pasture and reserve species biodiversity and crop rotations
- The total scores for Pasture Lands, Cropping Lands and Woodland Areas are expressed as a percentage to a maximum of 100%. If optimum farm management practices are implemented then the score is 100%, and the quantity of Soil Carbon Sequestered is calculated as the maximum amount possible for that specific soil type and climatic zone. The scores range from 15% to 100% under Pasture Lands, 8% to 100% under Cropping Lands, and 72% to 100% under Woodland Areas. These weightings are initial 'best estimates' for the different farming systems employed, and will be adjusted as better soil carbon data becomes available. The amount of soil carbon stored under the various land management practices are expressed as Total Soil Carbon Stored (tonnes/hectare) and (Soil Carbon %).

Farms were assessed and ranked using the FSCC spreadsheet and land management actions accurately mapped using GIS technologies. Importantly, farmers selected by this process received incentive funding, and have agreed to undertake detailed soil testing and monitoring so that this information can be used to further our knowledge of soil carbon sequestration on agricultural lands.

Conclusion

The FSCC spreadsheet has proven to be a useful tool for selecting leading farmers who are increasing soil carbon through improved land management practices. It is an integral component of the Central West CMA incentive project that rewards good farming practices, with estimated soil carbon levels linked to specific land management activities, soil type and climatic zones. The land management practices defined within the spreadsheet have been used in the development of a soil carbon matrix model (Murphy 2009), and the formulas can be modified in the future as more accurate soil carbon data is obtained.

In addition, the FSCC spreadsheet has been an important educational tool and has helped CMA staff and landholders increase their understanding of land management and soil carbon issues. The Central West CMA soil health program and the development of the FSCC spreadsheet have greatly increased the level of understanding of carbon farming principles throughout the Central West Region of NSW. The FSCC spreadsheet offers a simple and rapid method for determining soil carbon sequestered in agricultural landscapes, and could be adapted to other regions and farming systems.

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Knowledge review on land use and soil organic matter in South Africa

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Abstract
There is limited data on the soil organic carbon status of South African soils. The limited data is further fragmented and uncoordinated. This study aims to provide a review of the research done in this regard. Approximately 58% of South African soils contain <0.5% organic C, 38% contain 0.5-2% organic C, and 4% contain >2% organic C. There are large differences in organic matter content within and between soil types, depending on climate, vegetation, topography, and soil texture. Over grazing and the use of fire in rangeland management resulted in significant losses of soil organic matter. Dryland cropping resulted in significant losses of soil organic matter, but this was not always the case under irrigation. The restoration of soil organic matter takes place very slow with the introduction of conservational practices like zero, minimum, and mulch tillage or the reversion of cropland to perennial pasture. Changes in soil organic matter content was restricted to the upper 300 mm and in many instances to the upper 50 mm of soil. The extent of these changes is dependant *inter alia* on land use, soil type, and environmental conditions.

Key Words
Carbon, nitrogen, burning, management, grazing.

Introduction
Organic matter influences the properties of mineral soils disproportionately to the quantities present. It is a major source of nutrients and microbial energy, holds water and nutrients in an available form, usually promotes soil aggregation and root development, and improves water infiltration and water-use efficiency (Allison 1973). The organic matter content of soils, however, varies widely. This is because the organic matter content of a soil increased during its development until a maximum equilibrium is reached. Variations in soil forming factors experienced on a landscape scale and the interdependence of these factors contribute to the large variability for soil organic matter contents. The relative importance of the soil forming factors on soil organic matter are viewed as climate > biota > topography ≈ parent material > time (Stevenson and Cole 1999). The equilibrium of organic matter in soils is disturbed by human activities such as crop and stock farming. Agricultural activities of this nature may either increase or decrease soil organic matter content in the long term. In the majority of cases, however, it is the latter. The equilibrium organic matter content in agricultural soils is governed by the kind of land use and resulting land cover (Arrouays *et al.* 2001). This knowledge review deals with the spatial variability of organic matter in South African soils, the extent of soil organic matter depletion on account of different land uses, and will elucidate some results on the restoration of soil organic matter.

Results and discussion
The land type survey (Land Type Survey Staff 2003) described and analysed the morphological and chemical data of 2380 soil profiles throughout South Africa. Barnard (2000) used these data to produce a generalized organic C map for virgin topsoils in South Africa. The organic C content ranged from less than 0.5% to more than 4%. Only 4% of the soils contained more than 2% organic C, while 58% of the soils contained less than 0.5% organic C. The remainder of the soils contained 0.5 to 2% organic C (Scotney and Dijkhuis 1990). South Africa is therefore characterized by soils with very low organic matter levels. The distribution of organic C in the topsoils was largely related to the long-term average annual rainfall (Barnard 2000), presumably because rainfall plays such a dominant role in determining the biomass production of the native vegetation in the country.

For master horizons (MacVicar *et al*. 1977; Soil Classification Working Group 1991) the organic O horizons had the highest organic C (25.88%), but only four of these horizons were described. The A horizons had the next highest organic C (1.22%), decreasing to 0.54% in the B, 0.40% in the C, and 0.34% in the R horizons. Organic C in the E (albic) horizons (0.48%) was lower than in the B (0.54%) or G horizons (1.10%), but higher than in the C (0.40%) or R (0.34%) horizons. The distribution of organic C in the master horizons can be related to the various soil forming processes active in these horizons.

Organic C under savannah ranged from 0.18 to 4.86%, with a median of 1.28%, while under grassland it varied from 0.09 to 12.53%, with a median of 2.51% (Barnard 2000). The amount, distribution, origin, and stability of soil organic matter in *Acacia* and *Burkea* savannas were investigated by McKean (1993). Soil organic matter contained between 50 and 95% of the organic C in the two semi-arid savannas. In both savannas, soil organic matter decreased with depth. However, the horizontal gradient of soil organic matter from the sub canopy to open habitat differed between the two savannas. In the case of the *Acacia* savannah it decreased whilst in the *Burkea* savannah it remained constant. Trees and grass contributed to soil organic matter in the two savannahs in approximate proportion to their primary production with grasses contributing 76% and trees 24%. A 51% turnover in clay-associated organic C over a period of 26 years was observed suggesting that in sandy savannah soils, clay-associated organic C has a much faster turnover time than the hundreds of years quoted for temperate ecosystems. In both savannas soil organic matter and microbial biomass were well related.

Du Toit and Du Preez (1993) obtained good relationships between the organic matter contents of virgin orthic topsoils and their fine silt-plus-clay contents (organic C: R^2 = 0.83; total N: R^2 = 0.83), the aridity indices of the localities where they were sampled (organic C: R^2 = 0.81; N: R^2 = 0.79) and the products of aridity indices of the localities and the fine silt-plus-clay contents of the soils (organic C: R^2 = 0.88; total N: R^2 = 0.90).

A baseline study by Le Roux *et al*. (2005) on the soil organic matter and vegetation was done in the Weatherley catchment near Maclear before this Moist Upland Grassland was converted to commercial forest. Organic matter was quantified to a depth of 1200 mm in 27 profiles, representative of the soils in this 160 ha catchment. The total amounts of organic C and total N were, respectively 111.1 and 8.6 Mg/ha for the excessively drained soils, 85.1 and 6.6 Mg/ha for the moderately well drained soils, 97.0 and 7.2 Mg/ha for the very poorly drained soils, and 88.3 and 7.2 Mg/ha for the freely drained soils. The average oven dry, above-ground biomass yield for the grassland cover was 3400 kg/ha/yr. Carbon sequestration efficiency by the grassland in the catchment was estimated to be 2.1 kg C/ha/yr/mm rain. Using some assumptions, it was possible to estimate that the equivalent value for *Pinus patula* would be approximately 2.8 kg C/ha/yr/mm rain.

Soil C stocks to a depth of 500 mm in untransformed, indigenous veld ranged from 21 Mg/ha in the Karoo to 168 Mg/ha in thicket, while N stocks ranged from 3.4 Mg/ha in the Karoo to 12.8 Mg/ha in grassland (Mills and Fey 2004a). The mean soil C of 5.6% to 100 mm depth in thicket was approximately five times greater than expected for this semi-arid region. Removal of vegetation due to grazing or burning reduced soil C and N at all sites. Soil C under intact thicket was greater than at sites degraded by goats, viz. 71 versus 40 Mg/ha in the upper 100 mm layer. Restoration of the thicket could therefore potentially sequester about 40 Mg C/ha. Soil C under plant cover was greater than exposed soil in the Renosterveld (28 versus 15 Mg/ha) and in the Karoo (7 versus 5 Mg/ha). In the 0-100 mm layer of burnt than unburnt plots the average organic C and total N were respectively 12 and 13% lower, viz. 1.68 versus 1.90% organic C and 0.13 versus 0.15% total N (Mills and Fey 2004b). The differences between the burnt and unburnt plots were accentuated in the 0-10 mm layer, which were respectively 0.8 versus 2.7% for organic C and 0.07 versus 0.23% for total N. They concluded that the top 10 mm of soil, which they named the pedoderm, was therefore likely to have a disproportionate effect on ecosystem functioning.

Cultivation caused average total N losses of 55% in the 0-150 mm layer, 17% in the 150-500 mm layer, and 6% in the 500-1000 mm layer. Reversion to pasture appeared to restore nitrogen fertility in the topsoil where leguminous trees were present, but not in their absence because the average total N was only 13% less in reverted compared to uncultivated soils (Prinsloo *et al.* 1990).

The C and N concentrations of the plinthic soils declined rapidly with increasing time of cultivation to approach equilibrium after about 30 years, when the concentrations of C and N were reduced, by 65 and 55% respectively, when compared to grassland soils. These losses occurred from all particle size fractions, but the organic matter associated with clay was more resistant than that in the sand fractions. The organic matter attached to silt continued to be lost as cropping continued, probably due to wind erosion (Lobe *et al*. 2001).

In the Harrismith, Kroonstad, and Tweespruit ecotopes some lands, continuously cultivated for more than 20 years were converted to perennial pasture of different ages. The restoration of organic matter, studied on pasture lands 25 years or less old, showed that on average 25% of organic C and 20% of total N, lost during the 20 years or more of cultivation, had been restored (Birru 2002). Most of this restoration took place in the 0-50 mm layer, a little in the 50-100 mm layer, and very little in the 100-200 mm layer. Results showed a wide variation in the rate of organic matter restoration between sites in each of the ecotopes, due mainly to differences in natural resource factors and management techniques. Positive factors and techniques were: a favourable soil water regime, promoted by an adequate rooting depth of at least 500 mm; a clay content above 12%; gentle slopes; an aridity index above about 0.35; plant available nitrogen levels above 15 mg/kg; and the presence of a legume in the pasture. Burning negatively impacted on C and N restoration.

A study in northern KwaZulu-Natal compared cultivated fields to adjacent virgin soils and found that dryland sugarcane production depleted the average soil organic matter content from 3.87 to 3.31% in the 0-150 mm layer, from 3.33 to 3.19% in the 150-300 mm layer, and from 3.16 to 3.04% in the 300-450 mm layer (Van Antwerpen and Meyer 1996). The average depletion of soil organic matter under irrigated sugar cane was from 2.40 to 1.88% in the 0-150 mm layer, from 2.08 to 1.69% in the 150-300 mm layer, and from 1.46 to 1.39% in the 300-450 mm layer. Depletion of soil organic matter was therefore relatively higher in the irrigated than dryland areas. In both cases the depletion of soil organic matter decreased with depth.

In the Swartland near Malmesbury, Smit (2004) reported that after 11 years cropping systems had little influence on either organic C or total N in the 0-50 mm layer. In the 50-100 mm layer higher C and N were measured in the monoculture wheat plots than in the wheat-lupine-wheat-canola rotation plots. Deeper than 100 mm, especially total N and to a lesser extent organic C were, however, higher in the rotation than monoculture plots. Tillage practices, in contrast to cropping systems, had a large influence on both organic C and total N in the 0-50 mm layer. In this layer and to a lesser extent in the 50-100 mm layer organic C and total N increased as the intensity of tillage decreased from conventional clean tillage to no tillage. However, at 100-200 mm the conventional clean tillage plots had higher organic C and total N content than the conventional mulch tillage, minimum tillage, and no tillage plots. In general both organic C and total N increased in the 0-50 mm layer as the intervals between minimum tillage actions increased from every year to every fourth year.

Annually cultivated and permanent pasture soils had gained soil organic matter in the sandy, lower rainfall eastern Tsitsikamma, when compared to the undisturbed native vegetation (Milne 2002). At the higher rainfall, more clayey western end there was a loss of soil organic matter under both types of pasture in comparison to the undisturbed native vegetation. Soil organic C content was also lower under annual ryegrass than under permanent kikuyu pasture at all the sites reflecting the degrading effect of annual cultivation on soil organic matter.

A study by Du Preez and Wiltshire (1997) to establish organic matter changes as a result of crop production under irrigation focused on 21 farms at Ramah, Riet River, and Vaalharts irrigation schemes, with aridity index values of 0.13, 0.16, and 0.17, respectively. Average annual biomass production of the native veld in the vicinity of the irrigation schemes is usually less than 0.8 Mg/ha, while the biomass production of irrigated wheat on the schemes was more than 17 Mg/ha. Soil organic C increased at eight farms, decreased at nine farms, and was not affected at four farms. Similar changes in total N were recorded at these 21 farms. Neither cultivation nor irrigation history of farms, nor soil properties provided any obvious explanation for these contrasting findings. It is therefore likely that irrigation, with the associated increase in biomass production counteracted the declining effect of cultivation on soil organic matter.

Conclusions

The results on soil organic C in South Africa, reported here are based on a few uncoordinated studies, by a few researchers. There is therefore no data based on a systematic and geo-referenced sampling programme. The limited available data shows that South African soils are characterized by very low organic matter levels. About 58% of the soils contain less than 0.5% organic C, 38% of the soils contain 0.5 to 2% organic C, and only 4% contain more than 2% organic C. There are large differences in organic matter content within and between soil types, depending on climate, vegetation, topography, and soil texture. Degradation of rangeland on account of over grazing resulted in significant losses of soil organic matter and is mainly attributed to less biomass production. The use of fire in rangeland management similarly decreased soil organic matter because the plant litter is destroyed. Dryland cropping resulted in significant losses of soil organic matter, but this was not always the case under irrigation. The restoration of soil organic matter takes place very slow with the introduction of conservational practices like zero, minimum, and mulch tillage. Reversion of cropland to perennial pasture also resulted in disappointingly slow soil organic matter restoration. Increases or decreases in soil organic matter content manifested only in the upper 300 mm and in many instances only in the upper 50 mm. The extent of these changes is dependant *inter alia* on land use, soil type, and environmental conditions.

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Land Management Actvities to encourage farmers to increase Soil Carbon

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Key words
Carbon, Soil, land management, carbon sequestration, Central West NSW.

Abstract
To improve soil health of farming lands, the Central West Catchment Management Authority (CWCMA) is implementing a project titled "Land Management Activities for Increasing Soil Carbon". The project will engage landholders across the Central West of New South Wales to undertake specific land management activities to increase soil carbon on designated areas of their properties. These areas will be benchmarked for soil carbon levels and spatially mapped. Over time soil carbon levels will be monitored and any improvements linked to land management practices, soil type and climatic factors.
The data from this project will contribute to a platform for any future activities towards the development of a soil carbon matrix methodology to estimate soil carbon. This also has implications to providing base Australian data for potential soil carbon trading.

Introduction
To increase soil carbon farmers need to incorporate the following principles into their land management activities: -

- Increase biomass production for improved groundcover and a biological food source
- Reduce soil disturbance and compaction
- Balance soil chemistry and nutrition for optimum plant growth
- Increase pasture or crop perenniality and / or increase the rooting depth of annual plants such as crops
- Increase pasture and reserve species biodiversity and crop rotations.

These principles will improve soil carbon and the general soil health important for establishing a resilient farming system. The intent of the project is to identify and document land management activities being implemented, determine the impact of these systems on the five principles above, benchmark and monitor soil carbon levels over time. Any soil carbon improvements will be linked to specific land management activities, soil type and climatic factors. Incentive funding will be provided by the CWCMA for farmers to adopt improved farming systems on a sustained basis to increase soil carbon levels.

Figure 1. Map showing extent of Central Western NSW

Cropping Management: pasture cropping, (Warden *et al.* in press) cover cropping, no tillage, zero tillage, and control traffic activities to increase the amount **of organic**

Grazing Management: time control & rotational grazing practices, to increase the amount **of organic carbon in the soil.**

Figure 2. Examples of improved land management activities that increase soil carbon include:

Methods
The Central West CMA will target 50 farmers and graziers from each of the Tablelands, Slopes and Plains areas of the catchment who express an interest in adopting and implementing a land management activity on a designated area of their property to increase soil carbon. The process detailed below will call for expressions of interest from farmers who will be selected for an incentive payment of $3000. The rationale behind the incentive payment is based on achieving higher soil carbon sequestration rates in the wetter and cooler climates in the catchment. The $3,000 correlates to areas of land to be managed as follows:

- **Tablelands $30/ha** (recommended project area is equal to or greater than 100ha).
- **Slopes $20/ha** (recommended project area is equal to or greater than 150ha).
- **Plains $10/ha** (recommended project area is equal to or greater than 300ha).

How do they apply?
To be eligible for the incentive all farmers have to submit an Expression of Interest (EOI) form which outlines all relevant project standards to be met.
Once returned the entries will be prioritised for a farm inspection by a CMA officer.

Project Standards
1. *Project implementation time* – Landholders must be willing to keep a record of their land management activities on the designated area of land over the next 10 years.
2. *Soil Testing Requirement* – Landholders agree to GPS located soil test at the commencement of the project and again at 5 and 10 years. The soil test will benchmark the current soil carbon level and then quantify any soil carbon increases over time. McInnes Clarke and Chapman (2008).

How are projects assessed?
Expressions of interest will be assessed against the specific criteria specified below by a panel of three experts who assign a ranking based on the answers provided by the landholder. The criteria to be used are:

1. Maximum Biomass Production and Groundcover Retention (max score 20)
Aspects to be considered in these criteria are farming operations such as not burning or overgrazing stubble and pastures (Geeves *et al.* 1995, Murphy *et al.* 2003,). Burning releases greenhouse gases, exposes soil to erosion, (Packer *et al*. 1992) loss of soil moisture and depletes soil nutrients and carbon. As an example of how applications will be ranked is a cool burn just before sowing has a higher score than an early hot burn after harvest. However to gain a greater score all stubbles need to be maintained and not grazed with stock. To achieve this criteria agronomy cannot be ignored to maximise biomass production. Aspects such as those listed below should be implemented:

- Controlling weeds, pests and diseases with appropriate rotations and sprayings; and

- Optimising soil nutrients by the use of fertilisers (organic/inorganic) and legume rotations.
- Maximising soil water storage to depth for optimum plant growth (Water Use Efficiency) (Chan and Pratley 1998)

Another example would be the implementation of a time controlled grazing system as opposed to a set stocking or a slow rotational grazing system.

2. Increased Plant Rooting Depth (with perennials or healthier crops) (max score 20)
Enabling plants to root deeper in the soil will potentially build soil carbon to depth in the soil profile. Having perennial plants in a farming system has two advantages in producing more biomass for conversion to soil carbon by:

- Being able to respond quicker than annual species when moisture is available. This is particularly the case in summer and after dry periods when there is some green material for rapid photosynthetic activity when rains are received.
- The ability to store more carbon deeper in the soil as perennials are deeper rooting.

Deep soil carbon is also increased (but to a lesser extent) by increasing the rooting depth of annual crops (Jayawardane *et al* 1994) and pastures in healthy soils. This can be done by addressing surface and subsoil constraints and having adequate soil nutrition.

3. Increased Biodiversity by increasing the number of species (max score 20)
In diverse pastures and cropping rotations, diversity of plant residues in the soil leads to a more complex food chain (Drew *et al*. 2004) and better soil health for biomass production (Chan and Heenan 1993). As soil microbial communities become more diverse and complex, they become more stable and create a 'suppressive' environment for pests & weeds (Roget 2002). A greater diversity of perennial pastures increases the potential of more biomass production.

4. Reduced soil disturbance and compaction (max score 20)
This is achieved by reducing cultivation (ploughing) and stock (Packer et al 1996) and machinery compaction to a minimum when cropping, sowing pastures and grazing (Packer *et al.* 1992). Having a litter layer from stubble retention, dormant pastures and brown manures will reduce the compactive effort and soil disturbance (Packer 1998). Both of these are essential to increase soil carbon. Also compaction is reduced by adopting controlled traffic techniques when cropping and using rotational time controlled grazing techniques to retain a protective cover of pastures.

5. Balanced soil nutrition (max score 10)
Building soil carbon requires balanced soil nutrition to maximise plant growth and encourage healthy biological activity (Kirkby 2009). To ensure a balanced nutrition farmers and graziers should regularly test their soil and plant tissue and monitor crop / pasture production to identify soil nutrient deficiencies. Land management actions to address any imbalances include activities such as fertiliser application (organic/inorganic and macro and micro nutrients), legumes for soil nitrogen, lime to address acidity issues and applying ameliorants such as gypsum and lime to address surface sodic soils.

6. Reduced of Greenhouse Gas Emissions from the Farm (max score 10)
To acknowledge a total farm commitment to reducing gas emissions points are rated for the adoption and use of renewable energy sources and innovative ways of reducing emissions or storing carbon on their farms.
Property inspection prior to payment
Following an EOI cull, successful farms are visited by a CMA field officer to check standards have been met and whether the land management practices have been implemented on the proposed area. If standards have been met, the proposed area is mapped and recorded, monitoring sites established and an agreement signed once reviewed by the landholder.

Conclusion
From our practical and scientific knowledge of soils and farming systems we conclude that the 5 basic principles described in this paper must be adopted by land managers and farmers to improve soil carbon. The amount will be influenced by biomass production and maintenance, soil type and climatic factors.

The project in this paper outlines a funding incentive program to aid farmers to implement land management practices on a sustained basis to improve soil carbon. Improved soil carbon is analogous with improved soil health. Likewise improved soil health has the advantages of improved farmer resilience and profitability as well as improved environmental outcomes.

The results of this project after 12 months will be available for presentation at the congress. It will then be possible to refine model showing the potential increase of carbon in Central Western NSW predicted by (Lawrie *et al.* 2006).

An added outcome of this project is providing data to improve a proposed soil carbon matrix model to enable farmers assess their potential to improve soil carbon. (Murphy 2009). Also the project will train CMA staff in land management and soil carbon issues.

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Litter and Carbon Accumulation in Soils after Forest Restoration: the Australian Experience after Bauxite Mining

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Abstract
Soil is the primary store of terrestrial carbon, and is seriously disturbed by mining activities. Soil carbon exists in various forms that are functionally different and have contrasting residence times as part of the soil organic matter store. Here I explore the nature of soil carbon, from surface litter stocks to humified fractions, as measured from various rehabilitated (reforested) bauxite mined land across Australia. Litter in rehabilitated forests tends to accumulate to higher masses than in the surrounding native forests. This may simply be a function of extra litterfall during forest regrowth and higher stem densities than in the natural forests, or of lower decomposition rates. The higher litter stocks in restored forests are sometimes reflected in a higher carbon concentration in the mineral soil. However the type of carbon that accumulates in the mineral soils after bauxite mining may be primarily of particulate organic form that is not stable in the long-term and may readily mineralize to CO_2. Further research is required to establish the stability of carbon in the soils of rehabilitated forests and woodlands compared to their natural systems and the implication for carbon accounting and climate change.

Key Words
Carbon, litter, organic matter, carbon sequestration, mining, restoration, forests.

Introduction
Bauxite mining in Australia is a surficial, strip-mining operation that modifies large areas of land compared to most other types of mining. A typical bauxite mine in Australia will disturb around 500 hectares of *Eucalyptus* woodland each year. The preferred type of rehabilitation practice is integrated with mining operations and allows top soils and sub-soils to be utilised with little or no storage where possible (Koch 2007, Tibbett 2010). Despite this practice, organic carbon is lost from the soils between an undisturbed soil and a new rehabilitated profile (Banning *et al*. 2008; Sawada 1999; Schwenke *et al*. 2000a; Spain *et al.* 2005). This is likely due to mixing during stripping, stockpiling, replacement and ripping practices, as well as greater oxidation (to CO_2) through microbial activity as new surfaces are exposed in the soil through disturbance.

Interest in soil carbon is increasing as concern mounts about the steadily rising concentrations of CO_2 in the atmosphere; which has risen by 31% since 1750 from fossil fuel combustion and land use change (IPCC 2001, Lal 2004). There is a hope that soil carbon may have a role to play in sequestering greater amounts of atmospheric carbon, with a particular expectation for the restoration of degraded landscapes in this respect (Lal 2004; Lal *et al.* 2007). Carbon accounting is likely to become commonplace (e.g. Waterworth and Richards 2008), and tracking terrestrial carbon stores, their transformations and net balance with respect to the atmosphere may become routine for mining companies. The component of terrestrial carbon that is most difficult to assess in a manner that limits risk for markets (likely to trade in carbon) is the carbon held belowground. In this paper I will give a brief overview of soil carbon, its types and measurement and report on the current data on soil carbon after bauxite mining in Australia. Although plant roots play an important role in belowground carbon they will not be considered in detail here.

The landscapes mined for bauxite in Australia
Australia has three areas in which bauxite mining is active. These include Weipa and Gove around the gulf of Carpenatria (both in the far North of Queensland and the Northern Territory respectively) and Boddington, Huntley and Willowdale in the coastal south west of Western Australia (See Tibbett 2010). Former mines in the south west were also at Del Park and Jarrahdale (Figure 1). The natural vegetation is an open *Eucalyptus tetradonta* woodland, in the northern sites and Jarrah (*E. marginata*) forest in the Western sites. The soils are ancient highly weathered, lateritic and broadly classed as Kandosols or Oxisols. The

formation of bauxite (Al) ore has accumulated a few metres from the current land surface after millennia of eluviation from the upper soil horizons.

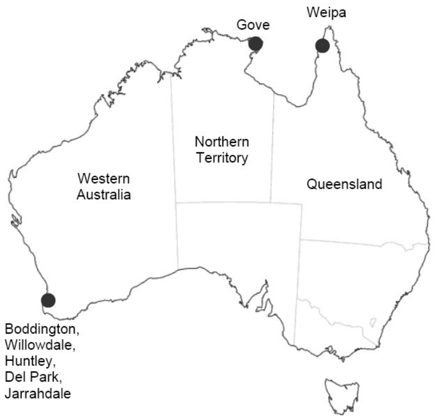

Figure 1. Location of bauxite mines in Australia where forest restoration has been implemented.

Carbon in post-mining soils

The primary source of soil organic carbon is from plant litter and this is related to the productivity of the vegetation biomass. In developing reforested systems, litterfall tends to be higher than in native stable systems. In bauxite mine rehabilitation, the density of seedlings that emerge and overall stem populations and tree basal area tend to also be higher than in the adjacent native forests. While belowground deposition is relatively difficult to estimate, aboveground litter stocks can be easily measured as a litter layer above the mineral soil; although the litter layer can sometimes be discriminated into upper and lower layers, with the lower layer having undergone greater comminution (Tibbett 2010). Stocks of litter on the forest floor are typically greater in restored sites than native, unmined forests sites. Litter accumulation has been reported as being almost four times greater (after 15 years of rehabilitation in Del Park Mine in Western Australia) than the surrounding Jarrah forest (Ward and Koch 1996). Litter masses that accumulated at two contrasting bauxite mines, one at Gove in the Northern Territory and the other at Boddington in Western Australia, are both greater in the rehabilitated than the native forests after around fifteen years (Figure 2). At Boddington, litter accumulation that matched the native systems took only ten years to occur while at Gove this took a little longer at 14 years. At the now closed Jarrahdale minesite (Western Australia) the amount of carbon held in the litter layer increased by 50% between 8 year old and 15 year old rehabilitated sites (Sawada 1999) This timeframe is in keeping with the natural senescence of many of the re-seeding, short-lived *Acacia* species that tend to dominate the canopy of the rehabilitated forests until this time. At Gove, for example, the *Acacia* tree basal area and the phyllode component of the litter tends to decrease from around 14 years (Tibbett 2010). From this, it seems likely that a rapid increase in litter stocks occurs as part of the natural successional cycle as (primarily legumous) re-seeders are replaced by the standard re-sprouting trees that make up the over-story in a mature forest canopy. Fire, an important aspect of litter mass balance in natural systems, has not been introduced into many of the systems reported here. This may have a profound effect on litter dynamics and the formation of inert organic matter; but will not be discussed further in this paper.

Litter is the feedstock for soil organic carbon and perhaps not surprisingly this affects the depth distribution of carbon in soils. Even within the top 10 cm of a natural (unmined) forest soil at Gove, carbon content declines steeply from the top centimetre of the mineral soil (ca. 2.6%) to the five to ten centimetre zone (ca. 1.2%) (Figure 3). This type of depth profile for carbon has been commonly reported for other soils in Australia (Spain *et al*. 1983). In the rehabilitated soils there is no depth discrimination of carbon in the earliest year and it takes five years to develop a measurable difference in the profile. As the age of rehabilitation progresses so does the discrimination of carbon percentage with depth. By year 14 this reflected the natural soils. After 14 years soil carbon concentrations increase to approximately double the values found in the natural soils. In such cases this represents a significant amount of extra carbon sequestered. In the sites that were measured between 20 and 26 years there is a notable decrease in the concentration of carbon in the soil and this may represent a stabilisation of organic carbon dynamics towards the native quotients. If this is so the long-term prospects for sequestering carbon in the soil may be no greater than the surrounding natural forests. Soil carbon, however, has a very complex chemistry and the total carbon measurements (based on wet oxidation or dry combustion) mask these details. Molecular analysis is possible using advanced analytical tools such as solid state nuclear magnetic resonance and pyrolysis followed by gas chromatography – mass spectrometry, that can be applied to characterise soil organic matter compounds in great detail. However, a useful intermediate technique is to characterise the organic carbon by particle size fractionations (Cambardella and Elliott 1992). In this method organic particle sizes are separated by wet sieving, and the size fractions are known to be related to the stability and recalcitrance. This in turn is related to the residence time of the organic carbon in the soil: the smaller the size fraction the longer the residence time.

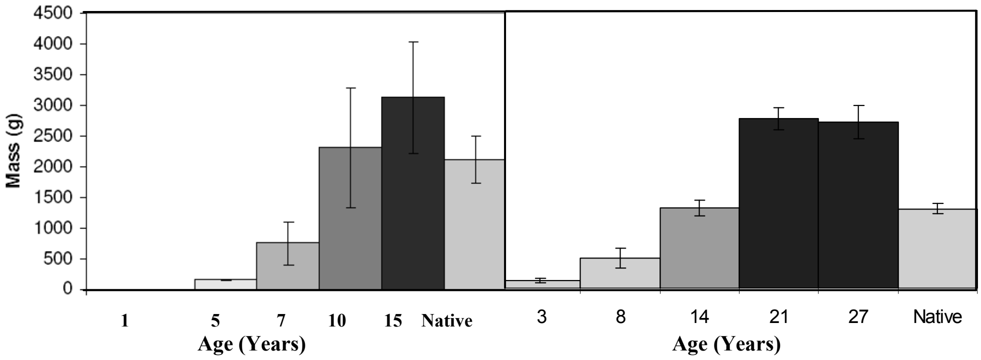

Figure 2. Surface litter masses (g/m^2) of land rehabilitated with *E. marginata* (Jarrah) forest at Boddington (left panel) and *Eucalyptus tetradonta* forest at Gove (right panel) and comparative litter masses on unmined (native) forest soil. Bars = SE. (After Tibbett 2010).

Various computer models attempt to determine the fate of these different fractions as soil carbon "pools". These models attempt to predict changes in soil carbon pools through time under different conditions. Their accuracy is limited but they provide a useful tool to anticipate the effects of changes in land use, management and climate. Their precision is partly governed by how well they can represent processes in soil and their calibration to real soils and conditions.

Some of these models have been commonly used in Australia and include RothC, CENTURY and Socrates (Kirschbaum *et al*. 2001). The century model tries to predict the active, slow and passive pools and has been used to model organic matter at the Weipa mine in Queensland (Schwenke *et al*. 2000b). The results of this modelling (for 100 years) showed that total soil organic carbon would increase steadily and that both active and slow pools would increase with time. The passive soil organic carbon, however, remained constant over this period.

Conclusions

The capacity of reforestation to accumulate carbon in soil, and sequester it there for the long-term, may be more limited than superficial measurements of carbon reveal. In studies outside the mining industry this has been shown to be the case (Arai *et al*. 2007; Jackson *et al*. 2002; Richter *et al*. 1999; Schlesinger and Lichter 2001) but the time frames and mechanisms needed to sequester recalcitrant carbon in the soil remain elusive. On this basis, there is clearly a need for more detailed studies on the soil carbon dynamics of post-mining reforestation schemes, both in terms of measurements of stocks, fractions and fluxes as well as predictive modeling.

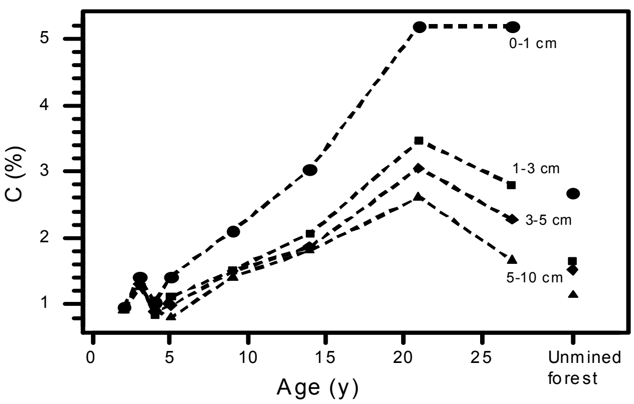

Figure 3. Carbon concentration at a range of depths in the upper profile in the soils of a 26 year chronosequence of rehabilitated sites (and in adjacent unmined native forest sites) at the Gove mine site (after Spain *et al.* 2005)

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N_2O and CO_2 Emission from Mine Soil Reclaimed with Organic Amendments

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Abstract
Surface coal mining causes drastic soil disturbance leading to serious degradation of inherent soil characteristics and productivity. The primary objective of mined land reclamation is to restore soils capacity to support vegetation and other essential ecosystem services. Organic amendments such as bio-solids, manure, composted manure and industrial by-products are effective in restoring soil productivity; however there has been little investigation of the potential for these amendments to give rise to potential green house gases such as nitrous oxide (N_2O) and carbon dioxide (CO_2). This laboratory incubation study compared N_2O and CO_2 fluxes from non-amended mine soil (control) and mine soil amended with lime and fertilizer, a manure (1.6g dry weight per 100g soil), composted poultry manure (7.1g dry weight per 100 g soil) and fresh poultry layer manure (1.6g dry weight per 100g soil) mixed with four rates of paper mill sludge (PMS) (0, 6.6, 9.9 and 13.2g dry weight per 100g soil) to achieve C: N ratios of 7:1,14:1, 21:1 and 28:1. The amount of N added with the compost and with all manure + PMS treatments was constant. The experiment was carried out at soil moisture contents of 60% and 80% water filled pore space (WFPS). The impacts organic amendments had on N_2O and CO_2 emissions were very short-lived. Combined application of fresh manure and PMS caused an increase in the N_2O and CO_2 emissions during the first 3 days of incubation. No differences were observed in emissions among the fertilizer, compost and only manures treatments.

Key Words
Nitrous oxide emission, mined land reclamation, manure, denitrification.

Introduction
Surface coal mining results in degradation of soil physical properties, significant loss of organic matter and nutrients and hence diminishes soil productivity (Akala and Lal 2001). Restoring the soil productivity and the establishment of sustained vegetative cover are primary objectives of mine soil reclamation. Use of post-mining landscapes for biomass production requires development of highly productive soils which can be achieved by large additions of organic amendments including agricultural manure. However, addition of large amounts of organic amendments results in significant nutrient loss unless measures are taken to stabilize the amendments and sequester the nutrients (Cravotta 1998; Sopper 1993; Stehouwer *et al.* 2006). Previous research has demonstrated that these leaching losses of manure N can be largely avoided either by composting the manure or by mixing the manure with high carbon materials such as paper mill sludge (PMS). However, these amendments could also create conditions conducive for nitrous oxide (N_2O) and carbon dioxide (CO_2) emission because of the presence of large amounts of organic carbon and nitrogen and thus increased microbial activity. Release of CO_2 and N_2O to the atmosphere represents loss of essential nutrients from soil and also emission of two major greenhouse gases. Nitrous oxide is a greenhouse gas 300 times more effective than CO_2 (Ramaswamy *et al*. 2001). To assess the potential for these amendments to cause such gaseous emissions, we conducted a laboratory incubation experiment using mine soil treated with various manure-based amendments.

Methods
Soil sampling
Soil for this study was collected from the surface layer (0-15 cm) of an active mine site in Clearfield Co., PA. Soil material was passed through a 2 cm screen to remove large rock fragments before being brought back to the laboratory. Soils were analysed for gravimetric moisture by drying for 48h at 105°C. Then the soils were sieved to 2 mm, were characterized for physico-chemical properties such as pH, acidity, Ca, Mg, K, P cation exchange capacity (CEC) and then they were stored at 4°C for further analysis.

Incubation experiment-Measurement of N_2O and CO_2 emission from soil
An incubation experiment was set up using 1L clear glass mason jars each containing 100 gm soil mixed with the amendments described in Table 1. Soils were adjusted at 60% and 80% water filled pore space

(WFPS). Soil oxygen (O_2) level is difficult to measure directly; therefore, changes in soil O_2 levels are usually described using a surrogate, such as water-filled pore space (WFPS) (Lemke *et al.* 1998; Davidson and Verchot 2000). Nitrification occurs at up to 60% WFPS (Davidson and Verchot 2000). Denitrification becomes dominant at WFPS 60% (Lemke *et al*. 1998) and at 80% WFPS, O_2 diffusion is restricted to the point where N_2O is used as an electron acceptor and reduced to N_2 (Veldkamp *et al.* 1998). However, these WFPS values are not exact limits for nitrification and denitrification, because O_2 availability is a combination of O_2 diffusion rate and O_2 consumption by heterotrophic activity (Ma *et al.* 2008). Moreover, maintaining the soils at these two constant moisture levels ensured that the soil does not become too dry or too wet because both these conditions can impede microbial activity. Each treatment at each moisture level was replicated 3 times for gas sampling and one extra replicate for measuring ammonium (NH_4^+) and nitrate (NO_3^-) concentrations in soil using KCl as an extractant. The jars were incubated at 20±2°C, were aerated on a regular basis and were checked for water content by weighing the jars one day before each gas sampling. Gas samples were collected at 0, 1, 3, 6, 10, 28 and 37 days and were analyzed for N_2O and CO_2 using a Gas Chromatograph (Varian CP-3800). The treatment effects were analyzed by using two -way ANOVA and single degree of freedom contrasts were used for planned comparisons using SAS (Statistical analysis software).

Table1. Quantities of amendment material, C and N added to mine soil

Treatment	Material added (g/100 g soil)	Total C added (g C/ 100 g soil)	Total N added (mg N/100 g soil)
Control	-	-	-
Lime and fertilizer	0.5 g lime + 0.07 g ammonium nitrate	0.6	24
Compost	7.1 g compost	2.45	121
Manure	1.6 g manure	0.65	121
Manure and PMS(14:1)	1.6 g manure + 6.6 g PMS	1.68	121
Manure and PMS(21:1)	1.6 g manure + 9.9 g PMS	2.52	121
Manure and PMS(28:1)	1.6 g manure + 13.2 g PMS	3.36	121

Results

Physico-chemical properties of soil

The physico-chemical characterictics of soil such as pH, acidity was measured. Soil pH was 4.5 and acidity 20.7. Also soils samples were analyzed for Ca, Mg, K, P and CEC. The CEC of the soil was found to be 17 meq/100 g of soil and Ca (190 ppm),Mg(106 ppm), K(67 ppm), P(7 ppm).

Nitrous oxide (N_2O)emission results

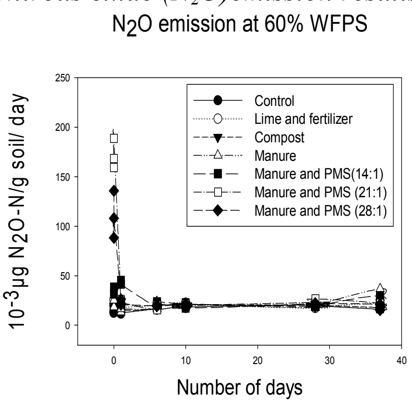

Figure1. Effect of organic amendments on N_2O emission from mine soil at 60% water filled pore space (WFPS)

Carbon dioxide (CO_2)emission results

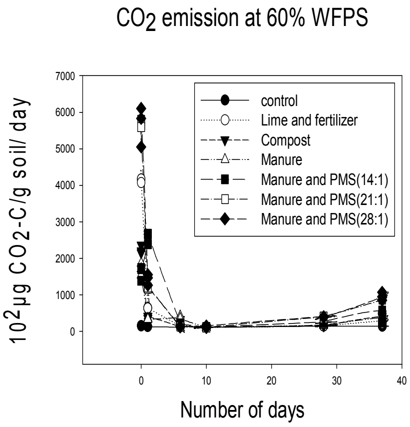

Figure2. Effect of organic amendments on CO_2 emission from mine soil at 60% water filled pore space (WFPS)

Discussion

Results indicated the effect of amendments and soil moisture content on N_2O and CO_2 emissions from reclaimed mined soils. Organic treatments, especially, the higher ratios of manure and PMS treatments (14:1 and 21:1 C: N ratios) gave rise to larger N_2O and CO_2 emissions from mined soils compared to untreated soil and soil amended with inorganic fertilizer. But the effect was limited to only first 3 days of incubation (Figure 1 and Figure 2). This increase of emission can be attributed to increased heterotrophic activity right after application of amendments. Increase of moisture content from 60 to 80% caused statistically significant increase in N_2O emission from manure based amendments (Figure not shown). In another study by Ciarlo *et al.* (2007), the greatest N_2O emission occurred at 80% WFPS where the conditions were not reductive enough to allow complete reduction to N_2. The NH_4^+ content in the KCl extract of soil remained high in all organic amendments but the level of NO_3^- was found to be low in all treatments except for the soil treated with lime and fertilizer. This can be due to lower level of nitrification conversion of NH_4^+ to NO_3^- by the nitrifying population in soil. The reason why nitrifiers were not active in soil could be related to soil pH (pH below 5) because in soil, autotrophic nitrifying bacteria grow best at neutral pH (Principles and applications of soil microbiology, 2nd edition). Therefore it can be summarized from the laboratory results that organic amendments have the potential to cause increased CO_2 and N_2O emissions. However, since these effects persisted only for a very brief period under controlled laboratory conditions, these results should be further tested *in situ*.

Conclusion

Over the past century, global average surface temperature has warmed by about 0.75ºC (Solomon *et al.* 2010). Much of the warming occurred in last half-century largely due to anthropogenic increases in well-mixed greenhouse gases (IPCC 2007). Therefore, care should be taken to avoid any human activity that would cause potential green house gas emissions capable of causing rise in earth's temperature to the environment. Hence, mine reclamation study needs to incorporate methods and develop strategies to come up with restoration mechanisms that are environmentally safe and at the same time are economically efficient. This study attempted to quantify and compare the emissions from mine systems reclaimed with different organic amendments. Once the results of this study are tested *in situ*, this study will be a useful addition to the growing body of literature focusing on restoration of degraded ecosystems. Data from this research coupled with studies focusing on biomass production in degraded lands will help develop a reclamation strategy that ensures optimum productivity while minimizing the emissions to the environment. Recommendations will be made to land managers on the basis of these research evidences to help them achieve specific reclamation targets with minimum environmental and economic costs.

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Potential change of soil carbon in Australian agro-ecosystems as affected by conservation management: data synthesis and modelling

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Abstract
Conservation agricultural practices (CAPs) have been suggested to be an effective way to enhance carbon (C) sequestration in agricultural soils to mitigate climate change. In this paper, we synthesised data from 49 peer-reviewed papers to examine the effects of adopting CAPs on soil C dynamics in Australian agro-ecosystems. We also used the Agricultural Production Systems Simulator (APSIM) to simulate changes in soil C under an annual wheat system as affected by nitrogen (N) application and stubble management across the Murray-Darling Basin (MDB). These results show that in Australian agro-ecosystems, cultivation led to C loss for more than 40 years, with a total loss of ~50% in the surface 0.1 m of soil. Adoption of CAPs generally increased soil C. Increasing crop frequency and perennial crops, when combined with stubble retention, had the greatest impact on increasing soil C, but the long-term impacts remain unclear. The same CAPs can have different outcomes on soil C under different climate and soil combinations. Modelling results using APSIM show that under an annual wheat system, 100% stubble retention and optimal N supply would lead to a basin-wide average increase in soil C of 8 Mg ha^{-1} to a depth of 1.5 m in 100 years. Under such a system, the rates of soil C accumulation increases along a transect from northwest to southeast of MDB. Fertilization and stubble retention must be combined in order to achieve the full C sequestration potential of the system. A modeling approach, combined with measurements, is an effective way to estimate the spatial distribution of soil C sequestration.

Key Words
Soil carbon sequestration, cropping system, tillage, stubble management, fertilization, climate.

Introduction
Cultivation of natural ecosystems has led to significant loss of carbon (C) from soil. Adoption of conservation agricultural practices (CAPs) has been widely recommended to enhance C sequestration in agricultural soils to mitigate climate change (Smith 2004; Lal *et al.* 2007). Lal (2004) estimated that the world cropland soils could potentially sequester 0.4-0.8 Pg C per year by adopting CAPs. This would correspond to 33.3-100% of the total potential of C sequestration in world soils. However, different CAPs will have distinct impact on soil C dynamics depending on soil, climate, and/or other agricultural practices (West and Post 2002; Chan *et al.* 2003; Christopher *et al.* 2009).

The impact of different CAPs on soil C has been extensively studied in recent years in Australian agro-ecosystems (Wells *et al.* 2000; Chan *et al.* 2003; Valzano *et al.* 2005). The most commonly held view is that CAPs lead to an increase in soil C. However, research results are inconclusive and vary depending on the specific CAPs and environmental conditions. For example, Heenan *et al.* (1995) showed that different agricultural practices caused significant differences in the soil organic C trend over 14 years, ranging from no change to an annual loss of 400 kg ha^{-1}. Several studies divided Australian agricultural areas into wetter (rainfall > 500 mm) and drier regions (Chan *et al.* 2003; Valzano *et al.* 2005), and found that whilst the adoption of conservation tillage increased soil C in wetter regions, it could not reverse the decline of soil C of croplands in drier regions. These inconsistencies in results may be due to the mix of different types of CAPs as well as the impact of different environments where the experiments were conducted. It is necessary to understand the effects of specific CAPs on soil C change, and how the interaction of climate, cropping systems and management strategies determine the net change in soil C in the different agro-ecological zones. In this paper, we synthesise the available information on soil C change following cultivation and the adoption of CAPs in Australian agroecosystem, and review the effects of adopting CAPs (i.e., the enhancement of rotation complexity, stubble retention and conservation tillage, and fertilization) on soil C content over time and space. Additionally, we use simulation modelling to investigate the potential capacity of soil to sequester C under different management scenarios and climate condition across the Murray-Darling Basin of Australia.

Methods and Materials

Synthesis of available data from literature

We focused on three major types of CAPs: i) cropping systems and rotation, ii) stubble management and tillage, and iii) the application of fertiliser. First, we extracted data from 49 peer-reviewed papers that studied the responses of soil C to adoption of CAPs in Australia. These data include the duration of experiments, soil sampling depth, soil C, soil type, and types of CAPs. For each CAP type and study, we calculated the relative change of soil C (C_r) under CAPs relative to conventional agricultural practices in paired experiments (i.e., other conditions and duration were kept similar), i.e., C_r = *(soil C under CAPs – soil C under conventional practices) / soil C under conventional practices x 100*. The relative change of soil C as affected by the duration of experiment and soil type were analysed for each type of CAPs.

Due to the complexity of the cropping systems in the selected studies, the CAPs involved changes of crop types and rotation types. We further divided them into three categories: 1) increased crop diversity (ID), referring to a change from continuous monoculture to continuous rotation, 2) increased cropping frequency (IF), a change from one crop per year to two or more crops per year, and 3) increased perenniality (IP), a change from annual crops to a rotation with perennial crops. To assess the effects of nitrogen-fixing plants on soil C content, we also separated rotation systems with and without legume crops (e.g., wheat-cotton rotations vs. wheat-chickpea rotations). Further, we analysed the soil C change between cropping systems with and without a fallow period.

The APSIM simulation

In order to examine how soil C changes in agricultural soils are affected by climate variation and management options, we used the Agricultural Production Systems Simulator, APSIM (Keating *et al.* 2003) version 6.0, to conduct long-term simulations. The model APSIM has been designed and developed for simulation of plant and soil processes by allowing flexible specification of management options. It provides the functionality to model soil C change as affected by environmental and management changes through its SoilN and SurfaceOM modules.

A simplified continuous wheat system was assumed to represent an annual cropping system, and one soil type with a plant available water holding capacity of 139 mm and total C content of 59 Mg ha^{-1} in the 1.5m soil profile was used in the simulation. The main focus was to quantify the impact of climate and managements (N application & stubble management) on relative changes in soil C. Seventy three sites, roughly uniformly distributed across the Murray-Darling Basin, were selected and historical weather data from 1889 to 2006 were obtained from the SILO database (http://www.longpaddock.qld.gov.au/silo/) and used in the simulations. Simulations were carried out to investigate the impact of nitrogen (N) and stubble managements on soil C. Three levels of N application rates (zero, 150 and 300 kg N ha^{-1}) and three stubble management options (100% removal, 50% removal, and retention) were simulated. For the 150 and 300 kg N ha^{-1} application, N application was split into two applications: 50 kg N ha^{-1} at sowing and the rest as a top-dressing at stem elongation stage. Stubble managements were implemented after harvesting. Soil C change under each management practice scenario was calculated as the difference of simulated total soil C in the soil profile (1.5 m) between 2006 and 1889. Multiple regression analysis was applied to assess the relationship between soil C change and climate conditions (i.e., mean annual rainfall and temperature).

Results – Synthesis of literature

Soil C loss after cultivation

Cultivation has led to a reduction in soil C in Australia. Combining data from 20 published studies that reported the C status in adjacent natural soils across Australian agro-ecosystems the result shows an exponential loss of soil C after cultivation, with most loss occurring in the first 10 years. A quasi equilibrium was reached after about 50 years of cultivation. The total loss of soil C in the surface 0.1 m was 51%, and the loss in the surface 0.3 m of soil was variable, ranging from 0.9 to 73.4%.

Soil C change after adopting CAPs

Cropping systems and rotation: Soil C content increased in the first few years after increasing crop complexity (i.e., ID, IP and IF) and remained stable thereafter. Compared with monoculture (with or without long-fallow) as the baseline, increased crop frequency (IF) and perenniality (IP) led to significant increase in soil C (10.1%, $F_{(2, 209)} = 22.72$, $P < 0.01$), while increased crop diversity (ID) only resulted in 5.3% increase in soil C. Introducing annual legumes into the rotation did not lead changes in soil C compared with nonlegumes. Introducing perennial plants into rotation caused the most significant increase in soil C, especially compared with continuous cropping with a fallow.

Soil C change after adopting different cropping practices varied with soil types. Crop diversity increased soil C content by 10% in Kandosols, 6% in Dermosols and 3.5% in Vertosols. Increasing crop frequency had the greatest impact on soil C content in Dermosols (56%, $n = 3$), while IP had the greatest impact on Vertosols (15.2%). On average, soil C content increased by 16.9% in Dermosols, which is markedly higher than the 8.5% and 10.3% increases in Vertosol and Kandosol soils, respectively ($F_{(2, 200)} = 2.99$, $P = 0.051$). However, different soils have distinct baselines of soil C content under the conventional agricultural practices ($C_{Conventional}$) and the limited datasets do not allow for detailed analysis to trace the exact causes for such changes.

Stubble management and tillage: Combining stubble retention and conservation tillage increased soil C content by 16.4% compared with stubble burning and conventional tillage. The impact of this combination was significantly higher than the impact of separate application of these two practices, with 5.7% for stubble retention only and 2.96% for conservation tillage only.

There is no apparent relationship between the magnitude of soil C change and the duration of both conservation stubble and tillage management. Both stubble retention and conservation tillage do not appear to lead to significant changes in soil C content in long-term (>30 years) based on currently available data. The effects of adoption of conservation stubble and tillage on potential soil C sequestration are dependent on soil types ($F_{(3, 273)} = 26.90$, $P < 0.001$). Soil C content increased by 26% on Kandosols, which is significantly higher than the 6.31% and 11.82% measured on Sodosols and Chromosols, respectively. Kandosols have a relatively higher baseline soil C content and the greatest increase in soil C among the four soil types. Vertosols show the lowest C increase of 3.3% following the adoption of conservation management. However, Vertosols have the highest soil C baseline, which makes the amount of soil C accumulation comparable with Sodosol and Chromosol.

Fertilization: In Australia, the response of soil C change to fertilization is largely dependent on available water supply to the crops. Higher N input occurs mostly in wet areas. After synthesising the data from 8 published studies on the relative change of soil C content ($F_{(4, 70)} = 16.27$, $P < 0.001$), the results indicated that, soil C increased with N input levels. However, the soil C did not continue to increase after the first several years.

Results - Modelled effects of fertilization and stubble retention on soil C and its dependence on climate
The APSIM simulation results show that in the Murray-Darling Basin, soil C would decrease rapidly under a continuous wheat cropping system if no fertiliser was applied or if all crop stubble was removed. At the basin-scale, if the crop growth is not limited by nitrogen stress (under optimal N supply, less than 150 kg N ha^{-1} for most cropping regions), at least half of the crop residue needs to be retained in the field in order to maintain the soil C level (to balance the C inputs with decomposition processes). Retaining all the stubble and optimal N supply would lead to a basin-wide soil C increase by around 8 Mg ha^{-1} in the upper 1.5 m of soil within 100 years.

The spatial pattern of the soil C change was significantly correlated with temperature and rainfall. Under the annual wheat system with 150kg N ha^{-1} and 50% stubble retention, the C accumulation rate increases along a transect from northwest to southeast of the basin. The simulated rate of change in soil C is negatively correlated with both temperature and rainfall in all simulation treatments, with the exception of high N (150 or 300 kg N ha^{-1}) combined with full stubble retention treatments, for which a positive correlation with rainfall was found. This implies that only under full stubble retention and optimal nitrogen supply can the soil C accumulation increase with productivity as a result of increasing rainfall.
Whether the agricultural soil is a C sink or source is dependent on local climatic conditions, cropping system and management strategies. Under the annual cropping system simulated with optimal N application, only about one third of the cropping regions were predicted to be C sinks. In contrast, if optimal N is applied and all the stubble is retained, more than 90% of the cropping regions may become C sinks.

Discussion and Conclusion
Soil C content decreased exponentially after cultivation in Australian agro-ecosystems. Although CAPs are considered to be effective to increase soil C, there are large variations in the effects of various CAPs over time and space. The review results show that only when increased crop frequency and perenniality are

combined with stubble retention can soil C be significantly increased. Other types of CAPs (e.g. no-tillage) had a lesser impact on total soil C change. Based on the available data, no consistent trend of increase in soil C can be found with the duration of CAPs applications. A question remains as to the long-term effectiveness of CAPs to sequester soil C. The impacts of fertilization are largely dependent on climatic regions and how the crop stubble is handled; a finding consistent with the APSIM simulation results. The modelling results show that the interaction of climate, soil, cropping and management systems determines C assimilation of the soil-plant system, the flux of C in and out of the soil, and thereby determines the net change in soil C.
In summary, the long-term impact of CAPs on soil C change is still inconclusive. Most of the studies are based on a limited number of experiments conducted at specific locations (climate and soil combinations) and in a relatively short period. Lack of explicit separation of different management options makes it difficult to analyse the impact of individual options. In addition, most studies are based on sampling obtained from only the top 20 cm soil and change in soil C in deeper soil are not considered. Due to the complexity of cropping systems and the large spatial variation of soils and climate, an experimental approach is always limited and impractical for spatial assessment of the soil C change region-wide. A system modelling approach based on sound understanding of processes in the soil-plant-atmosphere system provides an effective means to explore the impact of the complex interactions on soil C change.

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Secondary succession after fire in *Imperata* grasslands of East Kalimantan, Indonesia

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Abstract
We studied an early succession in *Imperata* grasslands in East Kalimantan, Indonesia, using plots that last burned 3 years, 4 years and 9 years previously on, secondary and primary forest. The species coverage data were analyzed using CANOCO software. While *Imperata* decreases, the average percentage of shrubs and young trees clearly increases with time. In the burned plots, *Melastoma malabathricum*, *Eupatorium inulaefolium, Ficus* sp., and *Vitex pinnata L.* strongly increase with the age of regeneration, but these species were rare in the secondary forest. Soils with more than 50% sand had a slower development towards secondary forest. The number of species was lower on the sandy soils, which showed a stronger increase with time of *Pteridium aquilinum* L.. This fern slows down subsequent vegetation development. Canonical correspondence analysis (CCA) of the environmental gradient and vegetation showed that pH, bulk density, sand and clay are the factors influencing the distribution of species. Canonical correspondence analysis showed also that soil properties had a strong influence on vegetation composition. *Melastoma malabathricum, Vitex pinnata* L.*, Lycopodium cernum, Vernonia arborea* Buch.-Ham., *Dicranopteris linearis* are all species associated with high levels of exchangeable Al and low pH.

Key Words
Early succession, Imperata grasslands, Pteridium aquilinum, soil properties.

Introduction
Kalimantan, the Indonesian part of Borneo, covers about 73% of the land area of the island, and has one of the important tropical forests in the world. Nowadays, large areas of primary forest in Kalimantan have been changed into secondary forest, oil palm plantation, timber estate plantation, slash-and-burn agriculture, and also grasslands such as *Imperata cylindrica.* When *Imperata* grassland are abandoned and not burned regularly, they will undergo a series of vegetation changes, a process called secondary succession. In the early phase of secondary succession, ferns, herbs, lianas and young trees (pioneer species) rapidly colonize the site. Leps (1987) mentioned that this early stage of succession influences the later stages of vegetation development, which in their turn determine the character of the secondary forest and the recovery of the original biodiversity. Soil properties also change during secondary succession. Upon burning, pH initially increases due to production of carbonates upon ashing of vegetation. Van der Kamp *et al.* (2009) described changes of soil carbon stocks under secondary succession, using the same plots as used in the present paper. The present paper describes the pathways of secondary succession in *Imperata* grasslands of East Kalimantan, Indonesia. The objectives of this study were (a) to examine how the species community develops after fire and whether different directions are observed; (b) to explore the relation between community structure and pattern and environmental gradients.

Methods
Data collection
All field data were collected in the area of Samboja Lestari (secondary succession) and Sungai Wain (primary forest) from January until April 2007. In total, 291 plots were analyzed of which 28 in Sungai Wain and 263 in Samboja Lestari. The dataset contains 19 transects with a length varying from 200 to 1000 meters. All the plots in Sungai Wain belong to a single transect. The number of plots per transect varied from 6 to 24 and distance between the plots varied from 2 to 150 meters. Vegetation was sampled in plots of 2*2 meter. The soil profiles were shallow and consisted of an A, an AB and a B horizon, to a maximum depth of 50 cm.

Data analysis and statistical methods
Changes in species distribution after last fire incidence were analyzed in spreadsheets of Microsoft Excel. For this purpose, data were transformed to percentage coverage. Canonical Correspondence Analysis (CCA) was applied to assess the relative importance of first and second major gradients of environmental variables in explaining the species distribution patterns. Eight properties of A-horizons were included in the analysis.

Results
In the whole study (BOS Samboja Lestari and Sungai Wain), 252 plant species were identified. Table 1 shows changes with time after burning of the cover of *Imperata cylindrica*, *Pteridium aquilinum*, and the percentage of shrubs and young trees. Table 1 indicates significant changes with increasing time of regeneration. After three years of regeneration, *Imperata cylindrica* had the highest average coverage; it becomes less dominant from the fourth year on. The average cover of *Pteridium aquilinum* is initially low but increases after 4 and 9 years of regeneration. Also the average percentage of shrubs and young trees clearly increases with time. In the secondary forest other tree species take over, and both *Imperata* and *Pteridium* have disappeared.

Table 1. The cover (%) of *Imperata cylindrica, Pteridium aquilinum*, and shrubs + young trees.

Regeneration time and number of observations	*Imperata cylindrica* (%)	*Pteridium aquilinum* (%)	*Shrubs and young trees* (%)
3 years (n=47)	63	10	21
4 years (n=94)	40	18	31
9 years (n=81)	18	25	44
Secondary forest (n=41)	0	0	30

Observation in the field suggested that sandy textures might influence the secondary succession. Soils with more than 50 percent of sand appear to have a slower development to secondary forest (Figure 1). Although there is little difference after 3 and 4 years of regrowth, after nine years shrubs reached higher cover percentages on less sandy soils and *Pteridium* on the sandy soils. Figure 1 show also that *Pteridium aquilinum* may induce stagnation in the regeneration of Imperata grasslands. *Pteridium aquilinum* can reach a height of 2-3 meters and casts much more shade than *Imperata* grassland. In addition, it has thick and deep rhizomes and slowly decomposable litter, which may impede germination of seeds from other species. This is consistent with Den Oden (2000), who mentioned that *Pteridium aquilinum* can induce stagnation in succession through shading, smothering, the build up of a deep ectorganic soil layer and the support of a high density of herbivore and seed-eating rodents.

Figure 2 show also that environmental variables have a strong influence on species composition. *Melastoma malabathricum, Vitex pinnata, Lycopodium cernum, Dicranopteris linearis, Syzygium lineatum, Vernonia arborea* are all associated with high concentrations of exchangeable Al and with low pH. The association of species such as *Melastoma malabathricum* with high concentrations of exchangeable Al or low pH values, was also mentioned by Watanabe and Osaki (2001) and Osaki *et al.* (2003). *Bridelia glauca,* and *Callicarpa longifolia* are associated with finer-textured soils, contrary to *Stenochlaeana palustris*, *Pteridium aquilinum* and *Spatolobus* sp., which are associated with coarser textures.

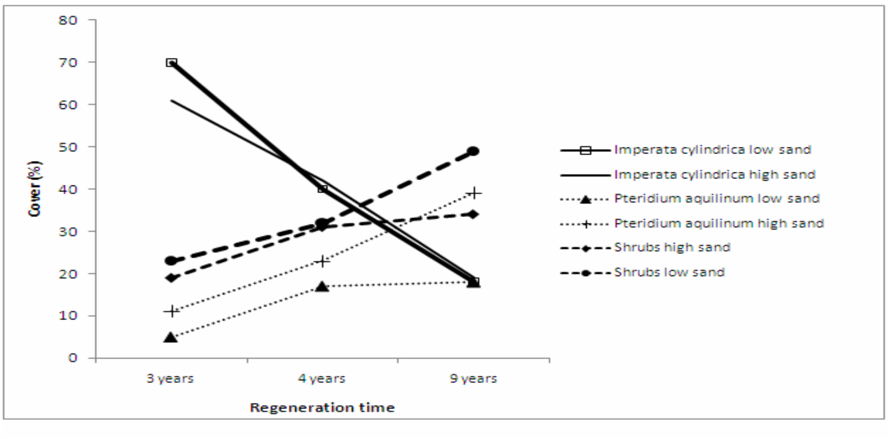

Figure 1. Cover (%) of *Imperata cyclindrica, Pteridium aquilinum* and shrubs + young trees in different phases of regeneration for high (> 50 %) and low sand content (< 50 %).

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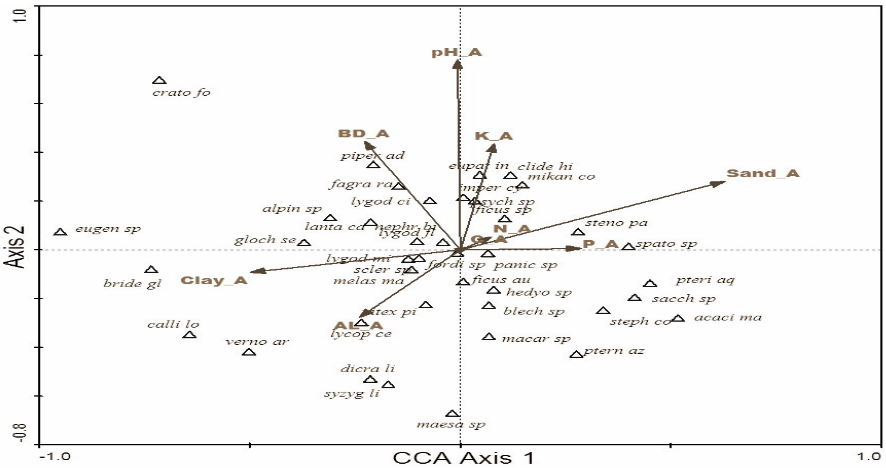

Figure 2. Plot of environmental variables and species (triangles) of *Imperata* grasslands systems (burned in 2004, 2003 and burned before 2003 plot on the first two CCA axes (only 40 species significantly abundant were analysed to CCA). Environmental variables in the A-horizon: (Clay_A=clay; AL_A=aluminium; BD_A=bulk density; K_A=Potassium; C_A= carbon percentage; N_A= Nitrogen total; P_A=Phosphor; Sand_A=sand). Variable vegetation: Acaci ma= *Acacia mangium ;* Alpin sp= *Alpinia* sp; Bride gl= *Bridelia glauca* ; Calli lo= *Callicarpa longifolia ;* Clide hi= *Clidemia hirta ;* Crato fo= *Cratoxylum formosum ;* Dicra li= *Dicranopteris linearis ;* Eugen sp =*Eugenia* sp. ; Eupat in= *Eupatorium inulaefolium ;* Fagra ra= *Fagraea racemosa ;* Ficus sp= *Ficus* sp. ; Ficus au= *Ficus aurata ;* Fordi sp= *Fordia splendidissima ;* Gloch se= *Glochidion sericeum ;* Hedyo sp= *Hedyotis* sp. ; Imper cy= *Imperata cylindrica;* Lanta ca= *Lantana camara;* Lycop ce= *Lycopodium cernum;* Lygod ci= *Lygodium circinatum;* Lygod fl= *Lygodium flexuosum;* Lygod mi= *Lygodium microphyllum;* Macar sp= *Macaranga* sp.; Maesa sp= *Maesa* sp.; Melas ma= *Melastoma malabathricum;* Mikan co= *Mikania cordata;* Nephr bi= *Nephrolepis biserrata ;* Panic sp= *Panicum* sp.; Piper ad= *Piper aduncum;* Psych sp= *Psychotria* sp.; Pteri aq= *Pteridium aquilinum;* Ptern az= *Pternandra azurea;* Sacch sp= *Saccharum spontaneum;* Scler sp= *Scleria* sp.; Spato sp= *Spatholobus* sp.; Steno pa= *Stenochlaeana palustris;* Steph co= *Stephania corymbosa;* Syzyg li= *Syzygium lineatum;* Verno ar= *Vernonia arborea;* Vitex pi= *Vitex pinnata.*

Conclusion

Imperata grasslands are not a final and stable stage of land degradation, but, when not maintained by frequent fires and human disturbances, regenerate spontaneously and rapidly to secondary forest. The introduction of native shrubs and trees will speed up this process. Recovery for agriculture has not been studied but should not pose major problems under management system without fire. *Pteridium aquilinum* may induce stagnation in the regeneration of *Imperata* grasslands.

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Soil carbon sequestration under chronosequences of agroforestry and agricultural lands in Southern Ethiopia

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Abstract
When forest is converted to other land uses, it often may lead to loss of organic carbon and nitrogen in terrestrial ecosystems. Soil organic carbon (SOC) and total nitrogen (TN) concentration and stocks, and the rate of change in the chronosequences of 12, 20, 30, 40, 50 years of agroforestry and agricultural lands in an Andic Paleudalf were investigated. A greater proportion of SOC and TN was concentrated in 0 to 20 cm depth and their concentration in agroforestry and agricultural lands was significantly lower than in the natural forest. The SOC stocks in all chronosequences of traditional agroforestry were higher than the corresponding chronosequences under agricultural lands. The loss of SOC stock under the chronosequence of 12 to 50 years of agroforestry and agricultural lands varied from 35 to 92 Mg/ha or 16 to 42 % of the stock under natural forest. The rate of SOC loss after 12 yrs of agroforestry was 5. 6 Mg /ha/yr, while it declined to 0.9 Mg /ha/yr after 50 yrs. The corresponding losses under agricultural lands were slightly higher. The trend of N losses, although of much lower magnitude (0.24 Mg /ha/yr after 12 yrs to 0.04 Mg /ha/yr after 50 yrs of agroforestry), was generally the same. These results show that the losses of SOC and TN stocks were higher in agricultural lands than in agroforestry but the stocks in both land uses increased with increasing chronosequences of 12 to 50 years.

Key Words
Soil organic carbon, soil nitrogen, SOC and TN stocks, agroforestry, agricultural land, soil profile.

Introduction
The major sources of the emission of greenhouse gases (GHGs) in addition to fossil fuel combustion are conversion of natural ecosystems such as forests and peat land areas to farm and other land uses (Kirby and Potvin 2007). In agricultural landscapes, agroforestry systems can be used as an alternative for economically sound and environmentally friendly land use approach.The C sequestration potential of agroforestry systems is estimated to be between 12 and 228 Mg/ha (Albrecht and Kandji 2008). Sequestering C in soils is often seen as a 'win-win' proposition; it not only removes excess CO_2 from the air, but also improves soils by augmenting organic matter, an energy and nutrient source of biota (Janzen 2006)

In the rift valley of Southern Ethiopia, extensive deforestation, overgrazing and conversion of natural ecosystem into arable land are rampant (Ashagrie *et al.* 2005). Remnant trees deliberately left from the clearance of the wood land and natural forest are scattered all over the agricultural lands. This is the local traditional (park land type of) agroforestry (AF) system prominent in the study area. Crops are grown during the rainy season in both the traditional agroforestry and agricultural lands.

It was reported that soil C and total N stocks in crop land soils was significantly lower than the soil C and total N stocks under natural vegetation in the southern highlands of Ethiopia but quantitative data that show the changes in soil C and N pools following the clearance of natural forest and the eventual land use change are rare. Thus the present study aims to: (i) to investigate the changes in SOC and TN stocks and their concentration under the chronosequences of 12, 20,30 ,40 and 50 years after conversion of forest to agroforestry and agricultural lands, (ii) to assess the distribution of SOC and T N in the soil profile (100 cm depth) ,and (iii) calculate the loss of SOC under chronosequences of these land uses.

Methods
Site description
Three sites namely, Ashoka, Leye and Beseko (7°17´N and 7°20´N and 38°48´E 38°49´E) in Gambo district of the Munessa shashemene forest enterprise of the south eastern highlands were selected. The altitude of the study site ranges from 2137 to 2215 m.a.s.l, with precipitation of 973 mm and maximum temperature of 26.6

^{o}C and minimum of 10.4^{o}C.The dominant forest species are *Podocarpus falcatus* Thunb.ex Mirb., *Croton macrostachys* Hochst.ex Rich., *Prunus africana* (Hook.F.) Kalkm. and *Schefflera abyssinica* (Hochst. Ex A. Rich). Soils of the study area are classified as Andic Paleudalf.

Soil sampling and analysis
Three adjacent land use types; natural forest, traditional agroforestry system and agricultural lands were selected. The later two land uses were of 12, 20, 30, 40 and 50 years of age after conversion from the natural forest. The sampling design followed was complete randomized design with four replicates and soil samples were collected at 0-10, 10-20, 20-40, 40-60, 60--100 cm depth increments from each replicate. The collected soil samples were air dried, grounded and passed through a 2 mm sieve prior to analysis. Separate core samples were drawn for bulk density determination. pH was measured by potentiometric method (Tan 2005) and SOC by titrimetric method (Walkley and Black 1934). Total N was measured using a LECO CHN-1000 Carbon and Nitrogen Analyzer

Measurements and calculations
Soil C stock (Mg/ha) was calculated by equation1.

$$\text{C stock (Mg/ha)} = BD * C_{conc.} * T * CF_{coarse} \qquad \text{(eq.1)}$$

Where Cconc.is carbon concentration (g/100g), BD is bulk density (Mg/m^3), T is depth thickness (m), and CF is correction factor (1- (Gravel % +Stone %) /100).

Statistical Analysis
The effect of land use, soil depth and other parameters were analyzed by the general linear model procedures of SAS. Multiple comparison of means for each class variable was carried out using the student-Newman-keuls (SNK) test at α =0.05.

Results

Distribution of SOC and TN in soil profile under the chronosequences of agroforestry and agricultural lands
In the 0-10 cm depth, SOC and TN under the chronosequence of 12, 20, 30, 40 and 50 of agroforestry and agricultural lands were significantly lower than that under natural forest ($P<0.0001$). However, SOC and TN under chronosequences of agroforestry and agricultural lands did not differ significantly. In lower depths (>20 cm) chronosequences did not show any significant difference with few minor exceptions.

Stocks of SOC and TN under agroforestry and agricultural lands
The SOC stock under natural forest and agroforestry chronosequence of 40yrs was significantly higher than that under the chronosequence of 12, 20, 30 years of both land uses ($P<0.0004$). The differences in the SOC stocks under the remaining land uses were not statistically significant. The SOC stock under all chronosequences of agroforestry was in general higher, although not statistically significant, than the corresponding chronosequences of agricultural lands (Table 1). The stocks of SOC and TN consistently increased with increasing chronosequence (from 12 to 50 years) of both land uses, suggesting that with time a new equilibrium in SOC sequestration was attained. The TN stock generally followed the same pattern as SOC. Similar to SOC, the average value of TN stocks under chronosequences of agroforestry were slightly higher than under those of agricultural lands.

Rate of losses of SOC and TN under chronosequence of agroforestry and agriculture
The loss of SOC stock under the chronosequence of 12, 20, 30, 40 and 50 years of agroforestry and agriculture varied from 35 to 92 Mg/ha or 15.9 to 41.5 % of that under natural forest, while the loss TN ranged from 2 to 5 Mg/ha or 11.1 to 27.8 %. Generally, the rate of loss of SOC stocks was higher for the first 12 to 20 years and it declined and approached a steady state in the chronosequences of 40 to 50 years under both land uses (Table 1). The rate of SOC was slightly lower under agroforestry (5.6 to 0.9 Mg /ha/yr) than under agricultural land (6.1 to 1.2 Mg/ha/yr). Similar to SOC, the loss of TN also declined with time. The rate of loss of TN was 0.24 Mg /ha/yr under the chronosequence of 12 yrs but it declined to 0.04 Mg/ha/yr at 50 yrs chronosequence and the corresponding decline under the chronosequence of agricultural lands were 0.32 and 0.07 Mg /ha/yr.

Table 1. Soil organic carbon (SOC) and TN stocks and the rate of change under agroforestry and agricultural lands after conversion from natural forest.

Land uses	SOC		TN	
	Stock (Mg/ha)	Rate of loss (Mg/ha/y)	Stock (Mg/ha)	Rate of loss (Mg/ha/y)
NF	221 ± 13.7 **a**		18 ± 2.2 **a**	
AF_{12}	154 ± 6.2 **bc**	5.6	15 ± 0.6 **a**	0.24
AF_{20}	156 ± 5.2 **bc**	3.2	16 ± 0.6 **a**	0.11
AF_{30}	144 ± 9.4 **bc**	2.6	15 ± 1.0 **a**	0.10
AF_{40}	186 ± 16.1 **b**	0.9	20 ± 3.3 **a**	-0.04
AF_{50}	174 ± 10.6 **bc**	0.9	16 ± 0.8 **a**	0.04
A_{12}	147 ± 8.4 **bc**	6.1	14 ± 0.6 **a**	0.32
A_{20}	129 ± 5.2 **c**	4.6	13 ± 0.7 **a**	0.24
A_{30}	137 ± 10.8 **c**	2.8	16 ± 1.7 **a**	0.06
A_{40}	173 ± 12.6 **bc**	1.2	18 ± 1.4 **a**	0.00
A_{50}	159 ± 10.3 **bc**	1.2	14 ± 0.7 **a**	0.07
	P=0.0001		P=0.1159	

Means followed by the same letter (s) in columns with stocks under land uses are not significantly different ($p \geq 0.05$)
AF= Agroforestry land use of 12, 20, 30.40,50 years and A= Agricultural land use of 12, 20,30,40,50 years of age after conversion of the natural forest

Conclusions

Conversion of natural forest into agroforestry and agricultural lands affected the SOC and TN negatively. Larger proportion of SOC and TN was concentrated in 0 to 20 cm depth and the concentration in this layer in both land uses was significantly lower than in the natural forest. In all chronosequences of agroforestry, SOC stocks were higher than in the corresponding chronosequences of agricultural lands but the SOC loss in the former land use was lower than in the later. The loss of SOC in both agroforestry and agricultural lands was many folds higher in the initial chronosequence of 12 and 20 years than in the later chronosequences of 40 and 50 years. Higher SOC stocks suggest that integrating more trees with proven multipurpose functions in all agricultural landscapes has a higher potential of sequestering SOC.

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Soil organic carbon dynamics in physical fractions in Black soils of Northeast China

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Abstract
It is important to identify the impacts of soil management systems on soil organic carbon (SOC) and especially SOC fractions that are active in soil aggregate stabilization. A total 32 pairs of non-cultivated and cultivated soil samples were taken to study SOC in physical fractions and their relationships in Black soils. Soil particulate OC (POM-C, >53 µm), mineral-incorporated C (MOM-C, <53µm) and aggregate-associated C were measured. The POM-C content accounted for 9.1% of total SOC in 0-30 cm layer of non-cultivated soil and the majority of this C was coarse POM-C (>250µm). Comparatively, cultivation led to greater decline in coarse POM-C than fine POM-C (53-250 µm). The MOM-C greatly declined with depth in non-cultivated soils ($P<0.05$), but this decline did notoccur in cultivated soils. There were significant positive correlations between the coarse POM-C (also fine POM-C) and total SOC, macroaggregate-associated C, respectively, in non-cultivated soil. For the cultivated soils the same correlations became weak; however, the relationships between POM-C and microaggregate-associated C became strong and the POM-C loss was at the same rate as macroaggregate-associated C loss. The correlations between MOM-C and microaggregate-associated C, and total SOC were greater for cultivated than for non-cultivated soils. Also, MOM-C loss was significantly related to aggregate-associated C loss and total SOC loss. We speculate that POM-C was released and mineralized during cultivation resulting in breakdown of macroaggregates into microaggregates, and thus protecting soil aggregation could play an important role in C sequestration.

Key Words
Black soils; soil organic carbon, particulate organic carbon; mineral-incorporated C, aggregate-associated C

Introduction
Aggregate dynamics has been suggested as a key factor controlling SOC dynamics (Denef *et al.* 2001). Hence, it is important to identify soil management systems that improve the build up of soil organic matter and especially the active organic matter in soil aggregate stabilization (Oyedele 1999). Soil particulate organic matter (POM>53 µm) is closely related to soil water-stable aggregates, especially macroaggregates (Angers and Mehuys 1988), and rapidly responds to agricultural managements. Soil mineral-incorporated organic carbon (MOM-C <53 µm) is closely related to soil organic matter accumulation and sequestration (Fang *et al.* 2007). The Northeast Plain, dominated by Black soils (Udolls, US Soil Taxonomy), is an important region of crop production in China. Intensive cultivation with improper management practices has resulted in serious soil loss and soil degradation. Our objectives were to study the changes in total SOC, POM-C, MOM-C and aggregate-associated C in cultivated and non-cultivated Black soils, as well as the relationships among them. Results will be valuable for evaluating the mechanism of SOC loss induced by cultivation of Black soil.

Methods
Study site
Soil samples were collected from the Black soil zone in Heilongjiang and Jilin provinces (Figure 1). Black soil is located in the temperate zone with a continental monsoon climate. The mean annual temperature varies between 0.5 ºC and 6.0 ºC, and the mean annual precipitation varies between 500 mm and 600 mm with more than 80% occurring in June to September.

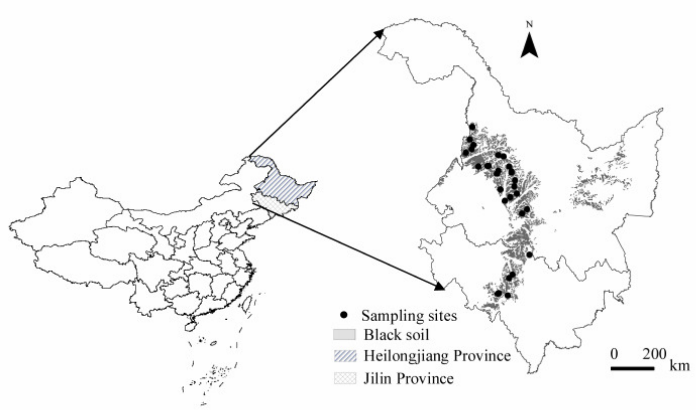

Figure 1. Sampling sites in Jilin and Heilongjiang provinces.

Soil sampling and analysis

In 2004 and 2005, 32 pairs of Black soil samples were collected from cultivated/non-cultivated sites. In each paired site, soils were developed under similar soil-forming conditions and the non-cultivated counterpart was never used for crop production. Soil samples were taken to a depth of 30 cm including 0-5, 5-10, 10-20, and 20-30 cm. Soil water-stable aggregates were measured using the wet sieving technique. For the remainder of this paper, the term "aggregate" means water-stable aggregate. Soil POM-C was dispersed by sodium hexametaphosphate solution, and transferred to a set of nested sieves having mesh sizes of 250 and 53 µm. Organic carbon remaining on the 250 µm sieve was termed coarse POM-C and that remaining on the 53 µm sieve was called fine POM-C. MOM-C was equal to total SOC minus POM-C. The SOC in bulk soil and fractions was measured using dry combustion.

Results

POM-C contents in cultivated and non-cultivated soils

POM-C in 0-30 cm layer of non-cultivated soils accounted for 9.1% of total SOC (Table 1). Coarse POM-C was greater than fine POM-C (Figure 2). POM-C was less in cultivated than in non-cultivated soils and C content was similar in coarse and fine POM. Reference to non-cultivated soil, cultivation decreased POM-C by 76.4%, 40.6%, 14.3% and 0.86% at 0-5, 5-10, 10-20 and 20-30 cm depths, respectively. The POM-C losses at 0-5 and 5-10 cm depths were much greater than total SOC loss (46.6% and 26.8%) (Liang 2008), suggesting sensitivity of POM-C to cultivation practices relative to total SOC.

Table 1. Comparisons of POM-C between non-cultivated and cultivated Black soils.

Soil depth （cm）	POM-C (g/kg)		POM-C/Total SOC (%)		（Non-cultivated-Cultivated）/Non-cultivated（％）	
	Non-cultivated	Cultivated	Non-cultivated	Cultivated	POM-C	Total SOC
0~5	9.17a(a)	2.16a(b)	20.5	9.0	76.4	46.6
5~10	3.42b(a)	2.03c(b)	10.6	8.6	40.6	26.8
10~20	1.94bc(a)	1.66ab(a)	6.8	7.3	14.4	20.6
20~30	1.16c(a)	1.15b(a)	5.0	5.6	0.86	11.2
Weighed mean	3.13	1.64	9.1	7.2	24.6	22.9

Means inside and outside the parentheses and in the same column and followed by the same letter are not significantly different at $p = 0.05$, respectively. The same applies to Figure 2.

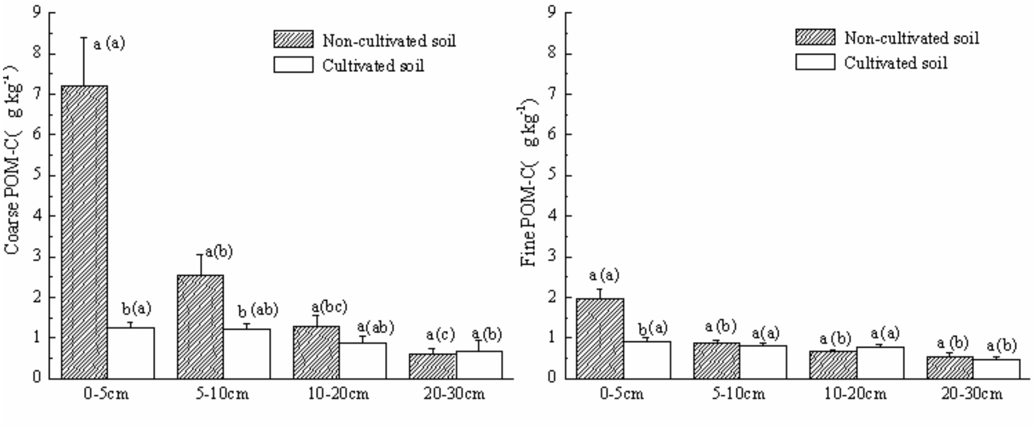

Figure 2. Coarse and fine POM-C in non-cultivated and cultivated Black soils (0-30 cm).

Changes in MOM-C

The MOM-C in 0-30 cm layer accounted for 90.5% and 92.5% of total SOC of non-cultivated and cultivated soils, respectively (Figure 3). Greater relative loss of MOM-C occurred in the surface (0-10 cm) than in sub-surface soil (10-30 cm). Although MOM-C was relatively stable and played an important role in maintaining C levels, the share of 86.6% total SOC loss from this fraction in cultivated Black soils indicates that attention needs to be paid to the fate of C in both fine and coarse particles when studying effects of agriculture on C dynamics.

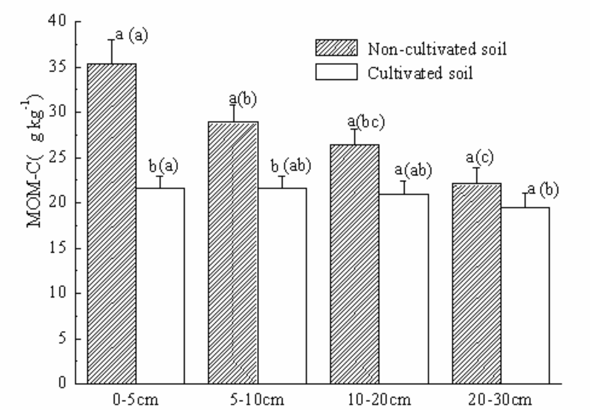

Figure 3. Soil MOM-C in non-cultivated and cultivated Black soils (0-30 cm).

Correlations among POM-C, MOM-C, aggregate-associated C and total SOC

There were significant positive correlations between coarse (also fine) POM-C and total SOC, macroaggregate-associated C, respectively, in non-cultivated soils ($P<0.01$) (Table 2). The similar relationships were weak for cultivated soils; however, the relationships between POM-C and microaggregate-associated C were stronger and POM-C (coarse and fine POM-C) was at the same rate as for macroaggregate-associated C loss. Also, MOM-C loss was significantly related to aggregate-associated C loss and total SOC loss. Therefore, POM-C and MOM-C both made large contributions to C accumulation retention in non-cultivated soil.

Table 2. Correlations between POM-C and aggregate-associated C, total SOC.

Non-cultivated soil	Coarse POM-C	Fine POM-C	MOM-C
SOC in >1000 µm aggregate	0.701**	0.525**	0.702**
SOC in 250-1000 µm aggregate	0.609**	0.533**	0.828**
SOC in 53-250 µm aggregate	0.170	0.314**	0.310**
Total SOC	0.771**	0.615**	0.969**
Cultivated soil			
SOC in >1000 µm aggregate	0.222*	-0.007	0.271**
SOC in 250-1000 µm aggregate	0.377**	-0.011	0.684**
SOC in 53-250 µm aggregate	0.228**	0.476**	0.677**
Total SOC	0.512**	0.292**	0.991**

* Correlation is significant at 5% level; ** Correlation is significant at 1% level

Table 3. Correlations among values of SOC loss for size fractions in cultivated soils.

	Macroaggregate-associated C loss	Microaggregate-associated C loss	Total SOC loss
Coarse POM-C loss	0.683**	0.043	0.695**
Fine POM-C loss	0.605**	0.169	0.690**
MOM-C loss	0.751**	0.378**	0.952**
Total SOC loss	0.834**	0.318**	1

* Correlation is significant at 5% level; ** Correlation is significant at 1% level

Conclusion

Carbon associated with soil macroaggregates (>250 µm) is more sensitive to agricultural management than POM-C. Cultivation weakens the relationship between the coarse or fine POM-C and total SOC, but strengthens the relationships between POM-C and microaggregate-associated C. Stronger correlations between MOM-C and microaggregate-associated C, total SOC in cultivated than in non-cultivated soils suggest that MOM-C could play an important role in C accumulation and retention.

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Soil organic matter stabilization in degraded semi-arid grasslands after grazing cessation

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Abstract
Soils in semi-arid grasslands are considered to store significant amounts of soil organic carbon (SOC) and to be of global importance for carbon sequestration. However, land use changes, in particular overgrazing, caused a large release of SOC in the last decades. The aim of this study was to investigate the sequestration potential of degraded grasslands after grazing cessation in terms of soil organic matter (SOM) stabilization. Intensively grazed as well as short- and long-term ungrazed grasslands were sampled in Inner Mongolia, Northern China, and analyzed for amount, spatial distribution and stabilization of SOM. Grazing exclusion led to a significant increase of SOC in the topsoil after three years and resulted in 35% higher amounts after 30 years. This increase is based on higher input of particulate organic matter (POM) and on increased amount of labile SOM physically protected within soil aggregates. This was evident as carbon mineralization of grazed sites with lower amounts of aggregate occluded POM was considerably higher compared to ungrazed sites. Analysis of the spatial distribution of SOM showed a heterogeneous pattern for ungrazed sites and a homogeneous distribution for grazed sites. Apparently, the recovery after grazing cessation starts with the formation of “islands of fertility”, where a higher input of water and organic matter promotes the development of vegetation and associated SOM patches. We conclude that grazing exclusion in degraded semi-arid grasslands has a high potential to immediately sequester atmospheric carbon and mitigate climate change.

Key Words
Soil organic carbon stocks, carbon mineralization, soil aggregates, steppe degradation, overgrazing, China.

Introduction
Degradation of semi-arid grasslands is a global environmental problem, particularly in Northern China, where 2.1 – 2.6 million km² are regarded as degraded steppe (Yang *et al.* 2005). In some regions of Inner Mongolia, up to 72% of the total area are classified as degraded steppe (Tong *et al.* 2004). The main cause of grassland degradation in Northern China is overgrazing as a result of increasing stocking rates after the change from a sustainable nomadic culture to a static grazing management in the last 50 years. Intensification of grazing was accompanied by a considerable loss of SOC in the grassland soils (Xie *et al.* 2007). Grasslands play a crucial role within the storage of global SOC containing approximately 15% of total SOC stocks (Lal 2004). The degradation of steppe soils can be attributed to lower input of organic matter (OM) into the soil as grazing decreases biomass input (Steffens *et al.* 2009b). Furthermore, animal trampling leads to soil compaction associated with a destruction of soil aggregates that in turn enhances soil erodibility. Formerly protected OM within aggregates is released and mineralized. In order to investigate the degradation of steppes and find solutions for a sustainable grassland management the sino-german research group MAGIM (Matter fluxes in grasslands of Inner Mongolia as influenced by stocking rate) was established in 2004. The study area is located in a semi-arid grassland in Inner Mongolia where continuously grazed plots as well as ungrazed plots, that were fenced in 1979 and 2005 were investigated. Our main research focus was the recovery of degraded semi-arid grasslands after cessation of grazing in terms of amount, spatial distribution and stabilization of SOM compared to continuously grazed steppes.

Material and methods
The study sites are located in the Xilin River Basin (43°38’ N, 116°42’ E, 1270 m a.s.l.) in the autonomous province Inner Mongolia, P.R. China. The region is part of the continental semi-arid grasslands of the Central Asian steppe ecosystem, with a dry and cold middle latitude climate. Mean annual temperature is 0.7 °C and mean annual precipitation is around 350 mm, with the highest values in the summer from June to August. The vegetation period from May to September is relatively short (<150 days). Zonal vegetation types are *Leymus chinensis* dominated steppe communities at areas with relatively wet soil conditions and

Stipa grandis dominated communities in drier regions. Soils are classified as *Calcic Chernozems* (IUSS 2006) which developed from aeolian sediments. They are characterized by a dark, carbonate-free Ah horizon followed by an Ach horizon with secondary calcium carbonate nodules.

In order to determine short-term effects of grazing cessation, a controlled grazing experiment was established at a *Leymus* dominated site in 2005. Topsoil samples were taken in 2005 and again in 2008 from ungrazed (UG), moderately grazed (MG) and heavily grazed plots (HG) with stocking rates of 4.5 and 7.5 sheep units per hectare, respectively, and analyzed for bulk density (BD), SOC, total nitrogen (N_{tot}), total sulphur (S_{tot}) and pH values. The effects of long-term grazing exclusion were investigated in detail for both steppe types at ungrazed sites that were fenced in 1979 (UG79) and compared to adjacent continuously grazed sites (CG) with a stocking rate of around 1.2 sheep per ha. For basic soil properties 3 soil pits were sampled at all study sites and analyzed for soil texture, BD, SOC, N_{tot}, S_{tot}, pH values and carbon isotope ratios ($\delta^{13}C$). To elucidate the spatial structure of selected topsoil parameters at the field scale, 100 grid points with spacings of 5 m and 15 m were sampled. For detection of small-scale variability at the plant scale, 40 randomly selected points were sampled inside areas of 2 m × 2 m at each plot. Semivariances were calculated for BD, SOC, N_{tot} and S_{tot}. To quantify the contribution of single SOC fractions to carbon sequestration a combined density and particle size fractionation was applied that separated bulk soil samples in POM and mineral fractions. Carbon mineralization was determined in an incubation experiment for a period of one month for UG79 and CG from *Leymus* dominated sites. For a detailed description of all investigations see (Steffens *et al.* 2009a; Steffens *et al.* 2009b; Steffens *et al.* 2008; Wiesmeier *et al.* submitted; Wiesmeier *et al.* 2009).

Results

Results from the controlled grazing experiment showed that SOC, N_{tot} and S_{tot} stocks calculated for the first 4 cm of the topsoil from all plots were comparable in 2005 with amounts of 1.0 kg/m^2 for SOC, 0.1 kg/m^2 for N_{tot} and 0.015 kg/m^2 for S_{tot}. In 2008, UG plots revealed significantly ($P < 0.05$) higher SOC, N_{tot} and S_{tot} stocks whereas MG and HG sites tended to show lower elemental stocks, but differences were not significant.

Table 1. SOC, N_{tot} and S_{tot} stocks determined in 2005 and 2008 for the first 4 cm of the topsoil of ungrazed (UG), moderately grazed (MG) and heavily grazed (HG) sites (standard deviation in parentheses, n = 40 for UG, n = 10 for MG and HG).

Site/year	UG 2005	UG 2008	MG 2005	MG 2008	HG 2005	HG 2008
SOC (kg/m^2)	1.04±0.23	1.11±0.24	1.03±0.14	0.88±0.05	1.01±0.19	0.81±0.09
N_{tot} (kg/m^2)	0.107±0.02	0.113±0.02	0.106±0.01	0.092±0.01	0.104±0.02	0.090±0.01
S_{tot} (kg/m^2)	0.015±0.0	0.016±0.0	0.015±0.0	0.013±0.0	0.015±0.0	0.013±0.0

Analysis of basic topsoil properties showed no significant differences for soil texture (sand 55–68%, silt 14–21%, clay 18–24%), pH values (6.6–6.9) and $\delta^{13}C$ (-23.3 – -24.0‰) between UG79 and CG sites. Stocks for SOC, N_{tot} and S_{tot} calculated from 100 sampling locations for each site revealed a significant increase (25–45%) for UG79 compared to CG from both steppe types. BD decreased after grazing cessation and was approximately 20% lower compared to CG. In general, *Stipa* sites showed significantly ($P<0.01$) lower stocks of SOC, N_{tot}, S_{tot} and higher BD than *Leymus* sites.

Table 2. SOC, N_{tot}, S_{tot} stocks, BD, pH (n = 100 for *Stipa*, n = 123 for *Leymus* CG, n = 98 for *Leymus* UG79) and soil texture (n = 3) of continuously grazed (CG) and ungrazed (UG79) *Stipa* and *Leymus* dominated steppe types.

Site	SOC (kg/m^2)	N_{tot} (kg/m^2)	S_{tot} (kg/m^2)	BD (g/cm^3)	pH ($CaCl_2$)	Sand (mg/g)	Silt (mg/g)	Clay (mg/g)
Stipa CG	0.72±0.10	0.08±0.01	0.009±0.0	1.35±0.04	6.9±0.3	676±11	145±6	179±6
Stipa UG79	0.98±0.16	0.10±0.01	0.012±0.0	1.10±0.08	6.9±0.2	642±31	180±24	178±6
Leymus CG	0.87±0.16	0.09±0.02	0.011±0.0	1.17±0.07	6.6±0.4	548±52	211±35	242±21
Leymus UG79	1.17±0.21	0.12±0.02	0.016±0.0	0.94±0.10	6.6±0.2	642±55	135±41	223±17

The spatial distribution of SOM at the field scale showed generally a more heterogeneous pattern for UG79 sites and a homogeneous distribution at CG sites. However, semivariograms for SOC, N_{tot} and S_{tot} indicated a high spatial variability at smaller scales <5 m. This was confirmed by the analysis of the spatial SOM patterns at the plant scale. All semivariograms of both UG79 sites revealed semivariances approximately one order of magnitude higher compared to grazed sites, pointing towards a much stronger spatial dependence.

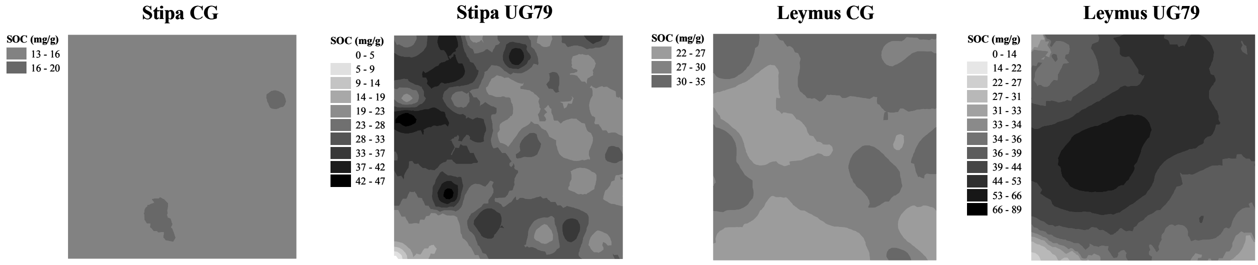

Figure 1. Spatial distribution of SOC at the plant scale (2 m × 2 m) of continuously grazed (CG) and ungrazed (UG79) *Stipa* and *Leymus* dominated sites.

The differences in terms of semivariance between grazed and ungrazed sites were more pronounced at the plant scale than the field scale. Ranges of spatial dependence at UG sites were about 23 cm for *Stipa* UG79 and 90 cm for *Leymus* UG79. Mean concentrations for SOC, N_{tot}, S_{tot} and BD determined at the plant scale for CG sites (40 sampling points) and for UG sites (20 points under bare soil areas and 20 points under vegetation patches) showed highly significant ($P < 0.001$) differences between vegetation patches and bare soil at both UG sites. CG sites and bare soil areas from UG sites revealed comparable values.

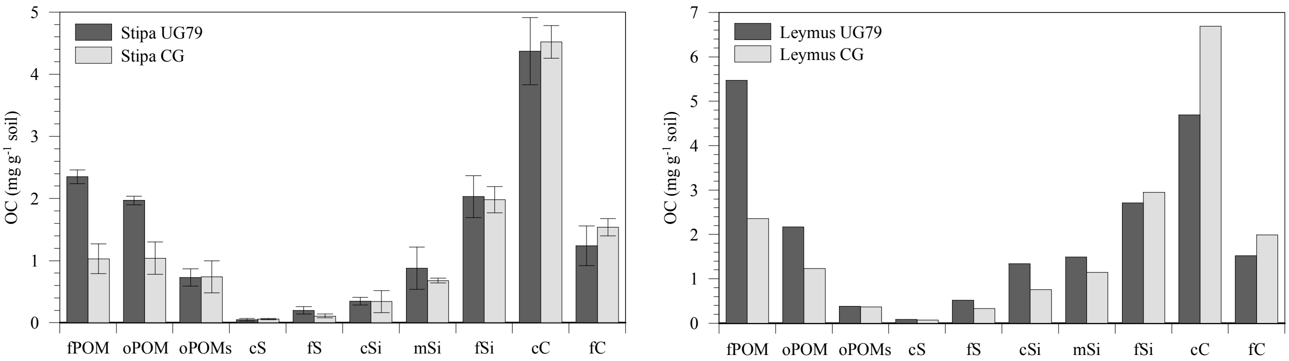

Figure 2. Contribution of OC from SOC fractions to the bulk soil of continuously grazed (CG) and ungrazed (UG79) *Stipa* (n = 3) and *Leymus* (n = 2) dominated sites (cS, fS, cSi, mSi, fSi, cC, fC = mineral fractions).

Results from the physical fractionation showed considerable higher amounts of SOC in free (fPOM) and in aggregate occluded particulate organic matter (oPOM) fractions for UG79 sites compared to CG sites. Grazing cessation had no effect on mineral fractions. Incubation of bulk topsoil samples from *Leymus* dominated sites showed a significantly higher carbon mineralization (+59%) for CG compared to UG79.

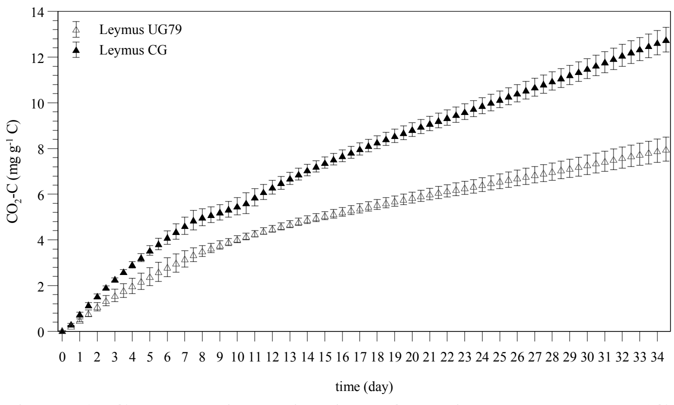

Figure 3. Carbon mineralization of continuously grazed (CG) and ungrazed (UG79) *Leymus* dominated sites.

Discussion and conclusions

Grazing exclusion for 3 years resulted in a significant increase of SOC (7%) (Table 1). Obviously, the recovery of degraded steppe soils starts with the cessation of grazing. After 30 years of grazing exclusion SOC stocks were approximately 35% higher for both *Stipa* and *Leymus* dominated steppe types compared to CG sites (Table 2). The ongoing carbon sequestration at long-term ungrazed grasslands indicates a high potential of degraded semi-arid steppes for mitigation of climate change after cessation of grazing. Increasing SOC stocks at ungrazed sites can be attributed to a higher input of organic matter into the soil as above- and belowground primary production is significantly higher there (Gao *et al.* 2008; Wiesmeier *et al.* submitted). This was confirmed by considerably higher amounts of POM at UG79 sites (Figure 2). The enhanced input of organic matter also promotes the formation of soil aggregates associated with a physical protection of labile SOC within aggregates. This was indicated by higher amounts of SOC in aggregate occluded POM at UG79 sites (Figure 2). Furthermore, a decrease of BD as a consequence of reduced mechanical stress was observed at ungrazed sites that probably supports soil aggregation. As a result of an enhanced physical proctection of SOM within soil aggregates, carbon mineralization is much lower at UG79 compared to CG (Figure 3). The increase of SOC after grazing exclusion starts with the formation of single resource patches. Analysis of the spatial distribution of SOC showed a homogeneous pattern for CG and a heterogeneous distribution for UG79. This patchy pattern can be explained by the development of "islands of fertility" under grass tussocks after grazing exclusion. This was apparent as the ranges of the spatial dependence of SOC were congruent with the extension of grass tussocks (Figure 1). Precipitation water as well as aeolian deposits are redistributed from bare soil areas with low infiltration to vegetation patches with higher infiltration. A higher input of water, which is the limiting factor in semi-arid steppes, and an increased accumulation of eroded materials at UG79 sites (Hoffmann *et al.* 2008) enhance primary production at single locations and create a heterogeneous pattern of vegetation and SOM. We conclude that these heterogeneous patterns are essential for carbon sequestration and productivity of semi-arid grasslands.

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Storing Soil Carbon with Advanced Farming Practices Central West NSW Australia - A Scoping Assessment of its Potential Importance

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Abstract
Improving and storing soil carbon is achievable in the Central West area of New South Wales Australia. This can be achieved with advanced farming practices which follow 5 basic management principles. However overriding factors in building soil carbon are:

1. The capacity of a soil to produce biomass which is determined by climate, the physical and chemical fertility of the soil, and land management practices.
2. Suitable soil conditions to encourage biological activity to convert organic matter into soil carbon.
3. Capacity of the soil to store soil carbon, as finer textured soils can store more carbon than sandier soils.

Using current knowledge on soil carbon levels and predicted soil carbon levels that are attainable it is estimated that 1110 M Mg CO_2 equivalents (CO_2-e) (400M Mg of soil carbon) can be sequested in the top 30 cm by 2030 in the Central West area. This is 10 times more than the estimated total greenhouse gas emissions per person in the Central West each year. The other major benefit of improved soil carbon is the improvement of soil health. Improving soil health has implications to improved soil moisture storage and retention, improved soil nutrition and more resilient plant growth for production. A critical factor in adaptation for any expected impacts of climate change. An added bonus is having resilient rural communities due to an improved socio economic environment.

Key Words
Carbon, Soil, advanced farming practices, Transeau's ratio, central west NSW.

Introduction
To increase soil carbon farmers there are five basic principles that need to be incorporated into existing or new farming practices:

1. Increase biomass production for improved groundcover and a biological food source
2. Reduce soil disturbance and compaction
3. Balance soil chemistry and nutrition for optimum plant growth and building soil carbon
4. Increase pasture or crop perenniality and / or increase the rooting depth of annual plants such as crops
5. Increase pasture and reserve species biodiversity and crop rotations.

The traditional farming practices which include practices such as overgrazing, hot burning of crop residues and excessive cultivation that do not incorporate these principles and are known to deplete soil carbon. New and more advanced farming practices such as stubble retention and zero tillage, cover cropping, time controlled grazing; pasture cropping, integrated nutrient management and increased use of perennial pasture species have the potential to improve soil carbon. Their impact on the sequestration rate and final level of soil carbon will depend on the level and number of the above principles implemented, climate and soil type. Implementing these principles will improve soil carbon and general soil health which is important for soil biological activity and creating a resilient farming system. There is however a higher order of factors that needs to be considered for soil biological function (Gupta 2009). These are:

1. Moisture
2. Soil pH
3. Temperature
4. Constraints such as pesticides and soil issues.

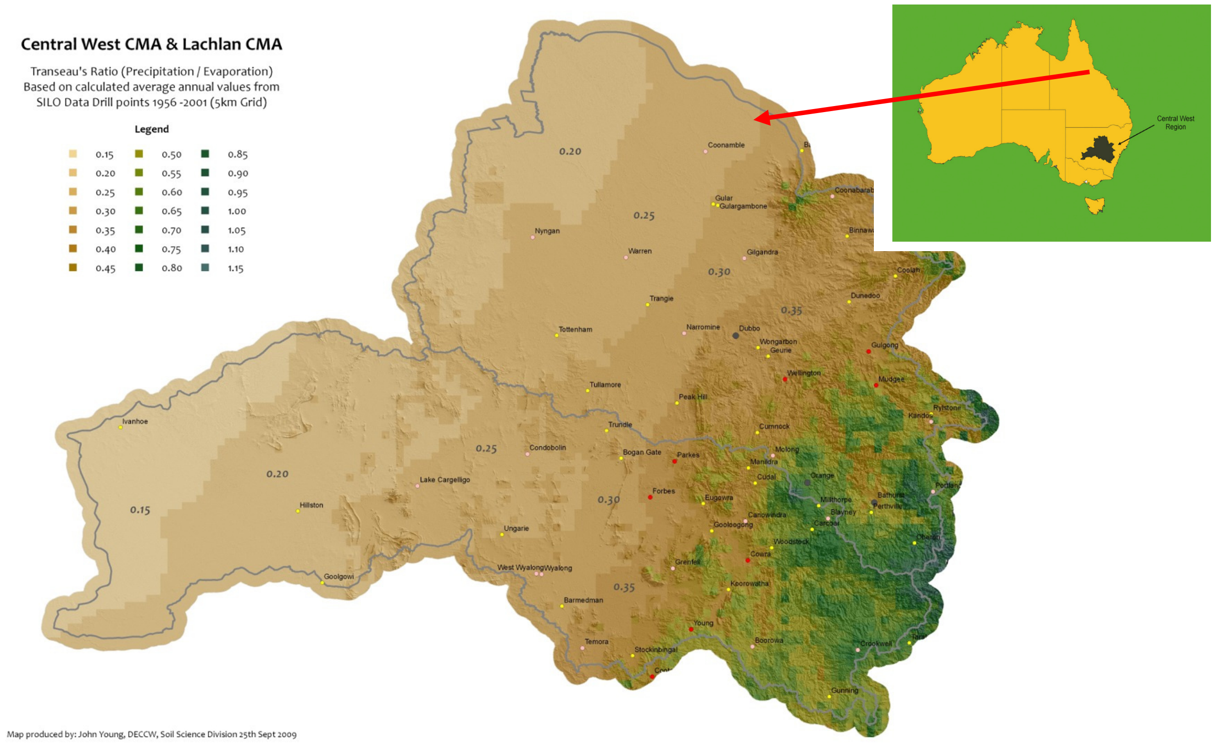

Figure 1. Transeu Map of Central Western NSW.

of these moisture and temperature are the main factors to consider in the Central West. The simplest way of delineating the climatic effect is using Transeau's ratio (P/E) for the area.

To calculate the potential of soil carbon levels for 2030, estimates were first calculated for the different Transeau zones (Table 1). These estimates included the broad soil groups of heavy textured and light textured (red soils).

The zones included the following estimated percentage area:

1. Rangelands
   1.1. Native vegetation with a good and bad cover – 50%.
   1.2. Annual and perennial pasture and bare soil – 50%
   Invasive native scrub was not included in the estimations as there has not been enough data collected on these areas
2. Plains
   2.1. Conventional and best cropping practices – 60%
   2.2. Annual and perennial pasture- 30%
   2.3. Native vegetation- 10%
   Regrowth of native vegetation was not included in the estimations as there has not been enough data collected on these areas
3. Slopes
   3.1. Conventional and best cropping practices – 30%
   3.2. Annual and perennial pasture – 60%
   3.3. Native vegetation – 10%
   Regrowth of native vegetation was not included in the estimations as there has not been enough data collected on these areas
4. Tablelands
   4.1. Annual and perennial pasture -70%.
   4.2. Native vegetation - 30%
   Regrowth of native vegetation was not included in the estimations as there has not been enough data collected on these areas

Table 1. Estimated current and maximum soil carbon storage due to improved practices for different climatic zones.

Zone (Transeau's Ratio)	Current Practice Soil Carbon Storage – forestry, native vegetation etc (Mg/ha/30cm)	Soil Carbon Potential with improved practices – forestry, native vegetation etc (Mg/ha/30cm)	Current Practice Soil Carbon Storage – Pastures (Mg/ha/30cm)	Soil Carbon Potential with improved practices – Pastures (Mg/ha/30cm)	Current Practice Soil Carbon Storage – cropping (Mg/ha/30cm)	Soil Carbon Potential with improved practices – cropping (Mg/ha/30cm)
Rangelands (<0.2)	13.5	22.5	9	15	N/A	N/A
Plains (0.2-0.25)	25	60	22.5	47.5	17.5	35
Slopes (0.25-0.35)	30	90	25	67.5	21.5	48.5
Tablelands (>0.35)	50	120	37.5	70	N/A	N/A

Using Table 1 and these area estimations, it is possible to further estimate the potential storage of soil carbon if landholders implement advanced agricultural practices.

However for the purposes of this scoping study the forest estimations have not been included as it is assumed there will be little change over the 20 year period. There is however data now emerging that we can improve the management and maybe the carbon levels in our current forests and vegetated areas. If this is the case then forest management can be included for improving soil carbon storage.

Assuming that only 80 percent of farmland will implement advanced farming practices implemented, this reduces the potential extra storage to 666 M Mg of carbon. Therefore the extra carbon stored is 666 M Mg minus 363 M Mg which is approximately 303 M Mg.

To be consistent with carbon discussions this is approximately 1110 M Mg CO2 equivalents (CO_2-e) of storage possible in the 20 year period between 2010 and 2030. To place this in perspective the following argument is presented:

1. The estimated 2005 NSW greenhouse emissions are 158.25 M Mg CO2-e for a population of 9 million people.
2. Concentrating this to Central West NSW with a population of approximately 295,000 people, this represents a proportionate emission of 5.17 M Mg CO2-e or 17.5 CO_2-e per head. This would be an overestimate considering the rural location.
3. Assuming emissions will be static over the next 20 years this is a total emission per head of 350 CO_2-e or 1.03 M Mg for the Central West.
4. Therefore with the implementation of advanced farming practices on 80 % of farmland in the Central West of NSW would sequester 10 times more carbon into the soil than the population are emitting.

It is again stressed that this is only a scoping investigation and the actual potential most likely will be less as current land use practices do not have such low carbon values and future land management practices cannot be expected to have such high values every where. More work is required to obtain more accurate soil carbon estimates.

These values compare favourably with an estimation by Lawrie et. al (2006) that farmers capable of storing at least 6 times more carbon than they emit. The estimated values in this paper are higher but are more accurate greater due to more data available on both soil carbon storage and emissions.

Table 2 - Potential to sequester soil carbon in the top 30 cm with advanced farming systems

Climatic Zone	Area (M has)	Land Use	Percent land use per zone (%)	Current Practice carbon storage (Mg/ha/30 cm)	Storage for each land use (M Mg/30cm)	Total carbon storage (M Mg /30cm)	Estimated Soil Carbon storage with improved practices (Mg /ha/30cm)	Potential soil carbon storage with improved practices (M Mg /30cm)	Total Carbon storage (M Mg/30cm)
Rangelands	1.25	Pastures	50	9	5.63	5.63	15	9.38	9.38
Plains	3.78	Pastures	60	22.5	25.52		47.5	53.87	
		Cropping	30	17.5	39.69	65.21	35	79.38	133.25
Slopes	9.66	Pastures	60	25	144.90		67.5	391.23	
		Cropping	30	30	62.31	207.21	48.5	140.55	531.78
Tablelands	3.22	Pastures	70	37.5	84.53	84.53	70	157.78	157.78
					Total	362.58			832.19

Conclusions

The authors appreciate there is a limitation of soil sequestration in that the soil carbon can only be sequestered once as a one way process. Once the capacity/saturation of the soil to store carbon is reached it cannot store any more carbon. This may be a long time for soils and the 20 years may be a conservative time frame. In the meantime the carbon dioxide emissions are a continuous process occurring every year but hopefully will significantly reduce in the next 20 years, the scope of this paper.

However the following conclusions can be made:

1. The estimations identify the potential for land management change to have a significant impact on storing soil carbon and carbon accounting. Also this soil carbon stored can make a significant contribution in the changeover period to lower emission technologies.
2. The study identifies the potential to use the 158.25 M Mg of CO2 (NSW annual emissions) as a standard unit to keep things in perspective.
3. There needs to be a refocus on implementing advanced farming practices that increase soil carbon and subsequently soil health rather than just productivity. Improved soil health will in the long term impact on maintained or improved productivity and lower inputs.
4. Future research should continue to benchmark and monitor soil carbon throughout the Central West region to obtain data on the impact of soil types, climatic zones and specific land use on soil carbon for accurate estimations of carbon sequestration for landholders.

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Using salt-amended soils to calculate a rate modifier for salinity in soil carbon models

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Abstract
In salt-affected soils, soil organic carbon levels are usually low as a result of poor plant growth; additionally, decomposition of soil organic matter may be decreased. Thus, the CO_2 evolution from salt-affected soils is likely to be lower than that from non-saline soils. Carbon models such as Rothamsted Carbon (RothC) that are used to estimate global CO_2 emission do not consider the effect of salinity on CO_2 emission. Given the large extent of salt-affected soils (19 percent of 20.8 billion hectares of arable land on Earth), this may lead to overestimation of CO_2 release. Two laboratory incubation experiments were conducted to assess the effect of soil texture on response of CO_2 release to salinity and to calculate a rate modifier for salinity in soil carbon models and study soil carbon dynamics: a sandy loam (18.8% clay) and a sandy clay loam (22.5% clay) in one experiment and a loamy sand (6.3 % clay) and a clay loam (42 % clay) in another experiment. Sodium chloride (NaCl) was used to develop a range of salinities viz. $EC_{1:5}$ 1.0, 2.0, 3.0, 4.0 and 5.0 dS/m. The soils were amended with 2% wheat residues and CO_2 emission was measured over 4 months. Cumulative CO_2-C/g soil decreased with increased salinity. Cumulative CO_2-C expressed as percent of the control soil (without salt addition) showed a lower impact of salinity on organic matter decomposition with increasing clay content. A decrease in particulate organic carbon (POC) associated with incubation was less in the higher saline soils whereas total organic carbon, humus-C and charcoal-C did not change over time and were not significantly affected by salinity. A significant exponential relationship was obtained between EC and the salt rate modifier, suggesting that a new salt rate modifier should be incorporated into RothC in order to accurately model CO_2 emissions from salt-affected soils.

Key Words
Carbon pools, respiration, RothC, salinity.

Introduction

As the global climate changes, it becomes increasingly important to understand how these changes will affect soils in general but also salt-affected soils which cover large areas in countries with dry climate such as Australia (more than 33% of the total area) . Salinity and sodicity are major constraints for successful crop production and have a large impact on soil organic carbon (SOC). Development and effect of soil salinity depends on many edaphic and pedological factors with soil texture being one of the most important. Clay soils have a greater EC buffering capacity; the higher the clay content, the higher the EC at which crop growth is negatively affected (Sumner *et al.* 1998). Soil organic carbon content is a function of C input and C turnover. With lower C inputs in saline soils as a result of poor plant growth, SOC stocks may be lower than in non-saline soils. On the other hand, a lower decomposition rate could lead to similar SOC stocks despite lower inputs. Turnover of SOC is mediated by soil microorganisms such as bacteria and fungi. In salt-affected soils, their activity can be decreased by osmotic stress and/or poor soil structure but little is known about the size and turnover of the different SOC pools in such soils. Several authors have studied the effect of salinity on soil carbon stocks and fluxes in short term incubation experiments. Both increased (Wong *et al.* 2009)) and decreased (Rietz and Haynes 2003) rates of soil organic matter decomposition with increasing salinity have been reported. An enhanced understanding of the implications of salinity on soil carbon dynamics is required to assess the implications of agricultural management on soil carbon stocks. Soil carbon models such as RothC have been successfully validated for non-saline soils (Smith *et al.* 1997) but do not consider the effect of salinity on CO_2 emission, an omission that we hypothesize will lead to inaccurate estimation of point, regional and global CO_2 emissions. In this experiment, different EC levels were imposed in four types of soil to address the following questions. What are the implications of soil salinity on CO_2 emission and SOC dynamics? Is the effect of salinity dependent on soil texture? Using the experimental data, can a rate modifier for salinity be introduced into the RothC model?

Methods

Soils

The study included two experiments with saline and non-saline soils from South Australia, each of which was setup as a completely randomised design with three replicates. One experiment was conducted with a sandy loam collected from Kadina and a sandy clay loam from Monarto, and another with loamy sand from Monarto and a clay loam from Kadina (Table 1). This classification is based on the USDA soil classification system.

Table 1. Physical and chemical properties of the soils.

Soil texture	$EC_{1:5}$ (dS/m)	Clay (%)	Bulk density (Mg/ m^3)	Water holding capacity (g/ g soil)	Total organic carbon (t/ha)
Loamy sand	0.08	6.3	0.13	1.66	55
Sandy loam	0.46	18.8	0.22	1.47	151
Sandy clay loam	0.82	22.5	0.34	1.41	112
Clay loam	0.30	42.0	0.42	1.28	58

Treatment of soils

Salinity was developed using NaCl to obtain an $EC_{1:5}$ of 1.0, 2.0, 3.0, 4.0 and 5.0 dS/m. The soils remained flocculated at the EC levels used in this experiment. The osmotic potential of the soil water was estimated using the equation: $O_s = -0.036\ EC_{meas}\ (\theta_{ref} / \theta_{act})$ (Richards, 1954), where O_s is the soil osmotic potential (MPa) at the actual moisture content (θ_{act} , g/g) of the soil and EC_{meas} is the measured electrical conductivity (dS/m) of the extract at the reference water content (θ_{ref} , g/g) of the 1:5, soil/water mixture.

Incubation and analyses

The air-dry soils were incubated for 14 days at 25 ^{o}C at 55 % water holding capacity (WHC) for the sandy loam, 50% WHC for the sandy clay loam, 75 % WHC for the loamy sand and 50%WHC for the clay loam to allow the microbial community to recover and stabilise. This soil water content was achieved by adding the appropriate amounts of saline solutions. Throughout the pre-incubation and the experiment, reverse osmosis water was added to maintain the required water content. After the pre-incubation, 2% mature wheat residue was mixed thoroughly with the soil. Twenty-five g of soil was transferred into PVC cores with a diameter of 3.7 cm and height of 5 cm with a nylon mesh base (0.75 µm, Australian filter specialist) and tapped to give the bulk density shown in Table 1. The cores were transferred into Mason jars fitted with stainless steel septum port to facilitate measuring of headspace gases. Headspace carbon dioxide (CO_2) concentration was measured throughout the course of the experiments (4 months) by using an infrared gas analyser. Using cumulative CO_2-C as an estimate, SOC pools (particulate organic carbon, humus-C, charcoal-C) and total organic carbon were determined by mid infrared spectroscopy three times, after 15-20%, 25-35% and 35-45% loss of POC.

Modelling of CO_2-C

The cumulative respiration data was used to develop an equation to modify the rate of decomposition according to salinity and the equation was incorporated into the RothC model (Jenkinson *et al.* 1987, Coleman and Jenkinson 1996).

Results

Cumulative respiration

Effects of salinity on decomposition of added wheat straw, and native organic matter, in soil depended on salinity levels and soil texture. Cumulative CO_2-C decreased with increasing salinity (Figure 1). The average difference in cumulative CO_2-C evolved between the control (original EC) and the highest $EC_{1:5}$ (5.0 dS/m) decreased with time; for example in the sandy loam soil the difference was 20.9% after 12 days, 9.6% after 76 days and 6.1% after 121 days. This may indicate adaptation of the microorganisms to salinity stress over time, although the decreased effect of salinity may also be due to the generally low respiration rates after 13 days. Cumulative respiration was higher in the sandy clay loam than in the sandy loam and higher in the clay loam than in the loamy sand (Figure 1). The decrease in cumulative respiration with increasing salinity (when expressed as $EC_{1:5}$) was higher in the coarse textured soils than in the fine textured soils. Because of the differential water content, the osmotic potential of the soil water at a given EC was lower in the finer textured soils. For example, osmotic potential at $EC_{1:5}$ 3.0 dS/m was -5.4 MPa for loamy sand, -4.8 MPa for sandy loam, -3.3 MPa for sandy clay loam and -2.6 MPa for clay loam . The relative decrease in cumulative respiration with increasing osmotic potential was similar in all soils.

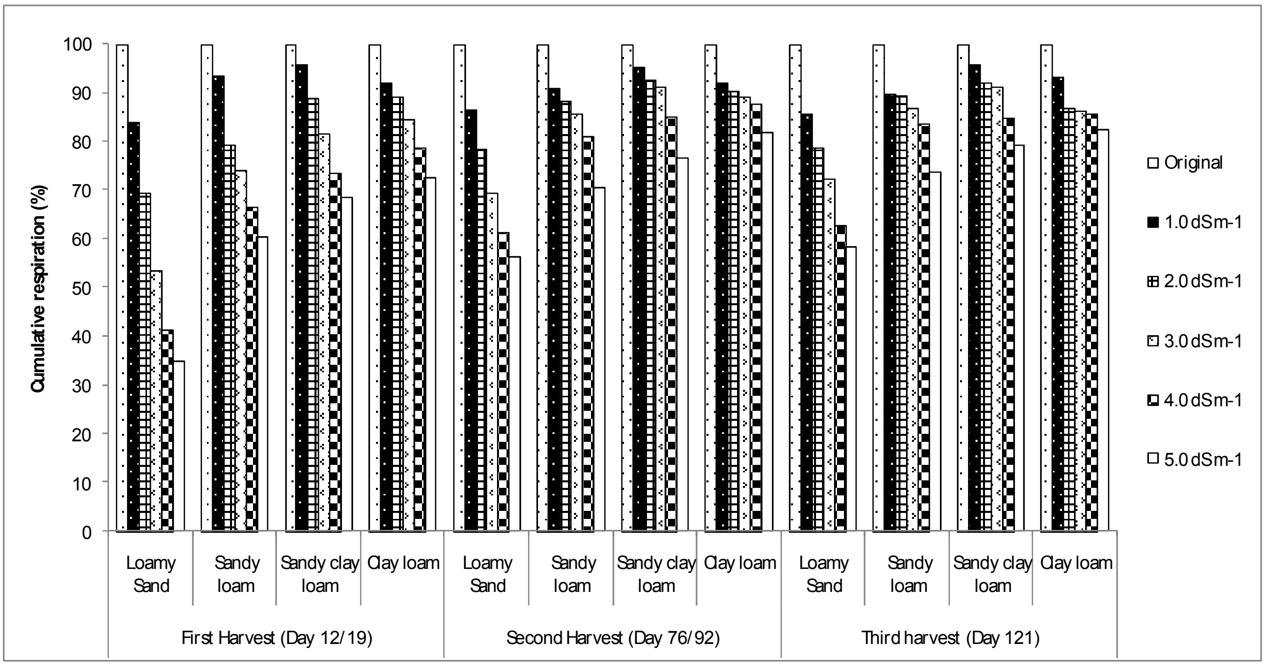

Figure 1. Influence of salinity ($EC_{1:5}$) on cumulative soil respiration (%) over 4 months in soils of varying texture.

Soil carbon pools

Due to reduced decomposition rates, a decrease in POC associated with incubation was less in the higher saline soils. Compared to the control soil (without added salt), the POC content at $EC_{1:5}$ 5.0 dS/m was 23% higher in the sandy clay loam at the third harvest. At a given EC level, POC significantly decreased with increasing time because of the loss of C as CO_2 as shown by a negative correlation between cumulative CO_2-C and POC. Humus-C and charcoal-C did not change significantly with salinity and /or time.

Use of RothC model to simulate CO_2-C in salt-amended soils

The Roth C was run to calculate the monthly soil CO_2 efflux from known total organic carbon content, clay content and laboratory conditions . The modelled CO_2-C was lower than measured CO_2-C of salt-amended soils. In order to match the measured CO_2-C, equilibrium and the short-term simulations of RothC were run with rate modifiers ranging from 0.2 to 1.0. The rate modifier for each salt-amended soil was calculated by a linear regression between measured minus simulated data and the rate modifier. The intercept of each equation was assumed to be a rate modifier for salinity (Table 2). A significant exponential relationship was obtained between EC and the salt rate modifier, suggesting that a new salt rate modifier should introduced into RothC to accurately model CO_2 emission from salt-affected soil.

Table 2. Calculated rate modifiers for salinity at various $EC_{1:5}$ levels.

$EC_{1:5}$ (dS/m)	loamy sand	sandy loam	sandy clay loam	clay loam
1.0	0.89	0.85	0.82	0.89
2.0	0.74	0.71	0.75	0.76
3.0	0.60	0.67	0.74	0.61
4.0	0.65	0.63	0.63	0.60
5.0	0.54	0.48	0.53	0.54

Conclusions

The lower sensitivity of respiration to salinity in the fine textured soils compared to the coarse textured soil is mainly due to the higher water content of the fine textured soils. We conclude that accounting for salinity will provide a more accurate simulation of CO_2 emissions from salt-affected soils but the modified RothC model will need to be evaluated against CO_2 efflux from naturally saline soils. This will help to improve understanding of turnover of soil organic matter as well as providing improved prediction of CO_2 emissions from salt-affected soils. In our experiments, the addition of salt may not have allowed the soil microbes to adapt to salinity as they would in the field where salinity develops more slowly. This may lead to an overestimation of the salinity effect. In order to test this, we are currently conducting an experiment with naturally saline soils.

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Management and Landscape Position Effects on Soil Physical Properties of a Coastal Plain Soil in Central Alabama, USA

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Abstract
Improved crop management is necessary due to raising production costs and environmental concerns. Input optimization from precision crop management might provide some solutions to these issues. Spatial variability of soil physical properties can significantly affect the implementation of precision agriculture techniques. A study was established in 2007 to determine the effect of management practices and landscape variability on soil physical properties (infiltration, aggregate stability and total C) of a 9 ha acre field located in the central Alabama Coastal Plain. The field was divided into three zones - summit, backslope and accumulation, using elevation, electrical conductivity and traditional soil survey data. Four management systems - conventional system with (CT+M) or without (CT) dairy manure, and conservation system with (NT+M) or without (NT) dairy manure – were established on a corn (*Zea mays* L.)-cotton (*Gossypium hirsutum* L.) rotation in 2001. Infiltration, aggregate stability and C content were generally lower in CT. Manure significantly increased the C content ($P \leq 0.001$), with 62% greater soil C content when manure was applied to CT, and 39% greater when applied to NT. Infiltration was greatest on the summit (14.5 cm/h), followed by backslope and accumulation zones (8.6 and 7.1 cm/h, respectively). No significant difference ($P = 0.69$ and 0.39, respectively) was found for aggregate stability and carbon between zones. Conservation tillage for 6 crop years thus far has improved infiltration and increased soil C content, whereas manure has only increased soil C content.

Key Words
Conservation agriculture, manure, soil physical properties, spatial variability.

Introduction
Soil physical properties affect water and chemical movement in the soil and can have a significant impact on crop productivity and the environment. Certain landscapes can have significant differences in soil physical properties due to spatial variability and can be the major cause of spatial variability in crop yields (Terra *et al.* 2005). Topography is a significant factor for soil differentiation (Jenny, 1941). Conventional tillage practices in areas with steep slopes can lead to erosion and soil degradation. Additionally, nutrient distribution within a soil profile can change with landscape position (Balkcom *et al.* 2005). Another important factor is the spatial distribution of soil C, since soil C can significantly affect soil chemical and physical properties. Landscape position plays an important role in C sequestration (Terra *et al.* 2005). Conservation tillage practices, such as non-inversion tillage (strip-tilling), can benefit production systems of southeastern United States. Conservation systems that include strip-till and winter cover crops can increase the soil organic C content and provide protective crop residue on the soil surface. Therefore, the objective of this work was to determine the effect of management practices and landscape variability on selected soil physical properties of a Costal Plain soil in Alabama, USA.

Methods
The study site was located in the Alabama Agricultural Experiment Station's E.V. Smith Research Center, near Shorter, Alabama, USA. Four management treatments were established in late summer of 2000 on a corn (*Zea mays* L.) and cotton (*Gossypium hirsutum* L.) rotation that had both crops present each year. The management systems included a conventional tillage system (chisel- followed by disc-plow) with (CT+M) and without (CT) manure, and a conservation tillage system (non-inversion tillage) that incorporated the use of winter cover crops with (NT+M) and without manure (NT). A mixture of rye (*Secale cereale* L.) with black oat (*Avena strigosa* Schreb.) was used as winter cover before cotton, and a mixture of crimson clover

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(*Trifolium incarnatum* L.) with white lupin (*Lupinus albus* L.) and fodder radish (*Raphanus sativus* L.) was used as winter cover before corn. Four strips per crop with an average length of 244 m were established across the landscape, with each strip having one of the four management systems. Each strip was further divided into cells to simplify sampling and field measurements. A total of six replications were established on the 9 ha field, with one replication consisting of eight strips (four management systems x two crops). Maximum slope was 8% with 9 soil map units are contained within this landscape.

Prior research work at the same field site delineated four distinct zones using a digital elevation map, electrical conductivity survey, and traditional soil mapping techniques. For this study, three of these zones were selected and recognized as summit, backslope, and accumulation zones in the landscape. Two cells per management and zone were selected to conduct soil physical properties characterization (Figure 1).

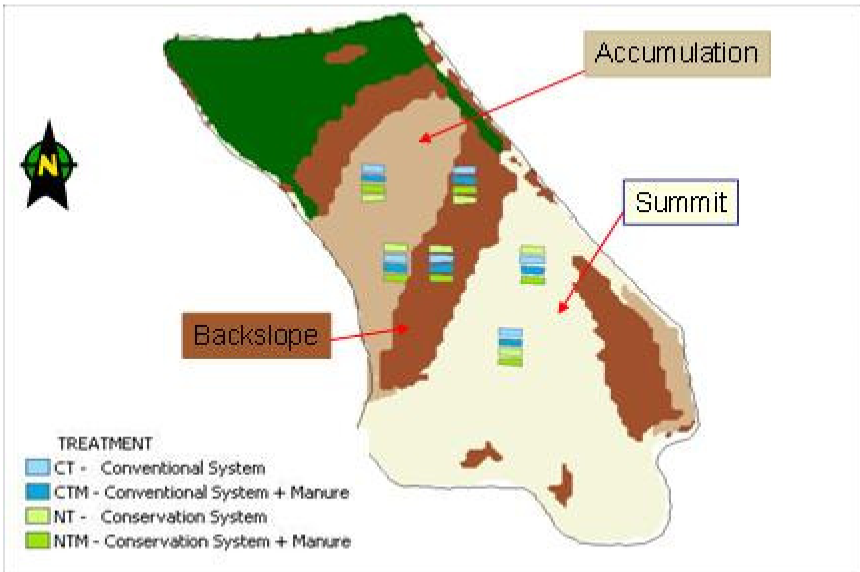

Figure 1. Location of sampling cells for each of the three landscape zones used in this study. The green region in the northern section of the field is an intermediate zone not included in this research.

Soil properties studied included total soil C by dry combustion at three depths, water infiltration with a mini-disk infiltrometer, and water stable aggregates (Nimmo and Perkings 2002). Other data were collected, including soil bulk density and water retention, but will not be presented here. Data were analyzed with the MIXED model procedure in SAS (SAS Institute Inc., Cary, NC). Management system, landscape position, depth, and their interactions were considered as fixed effects.

Results

On the surface 5 cm of soil, total C was greatest in the NT+M followed by CT+M, NT, and CT (Figure 2). Differences in C content between CT and NT were significant at the 0-5 cm of depth only. Non-inversion tillage increased C content by 54.7% on the surface soil, and by 1.3% from 5-10 cm of depth. However, C content was 2.5% lower in the NT than in the CT at the 10-15 cm depth. This lower C content can be attributed to the lack of soil mixing in the NT system. Nevertheless, soil C accumulation is greater with NT since C is broken down by increased soil respiration from CT operations. Small differences in C were observed with depth in CT, with C content ranging from 0.54 to 0.43%. All management systems had significant interaction ($P \leq 0.001$) with depth, except CT. The lack of difference in soil C content with depth in CT can be attributed to low C additions, greater C breakdown, and mixing of the surface soil (Figure 2).

Manure application significantly increased C content for CT+M and NT+M when compared to CT and NT on the top 10 cm of soil (Figure 2). Carbon content was increased by 81.9, 65.7, and 26.2% from 0-5, 5-10 and 10-15 cm of depth, respectively, when comparing CT and CT+M. A similar trend was observed for NT and NT+M, with C content increasing by 71.8, 5.7, and 4.2% for 0-5, 5-10 and 10-15 cm of depth, respectively. Landscape position had no significant effect ($P = 0.39$) on soil C content (Figure 3).

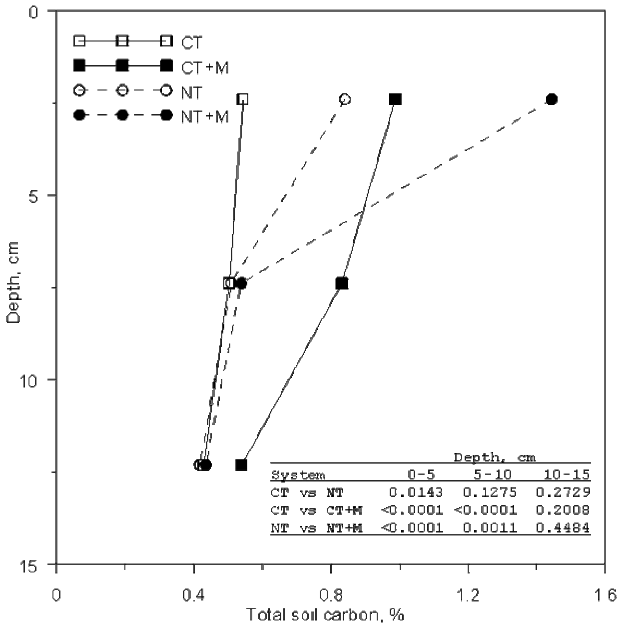

Figure 2. Total soil C content for the conventional (CT), conventional with manure (CT+M), no-till (NT), and no-till with manure (NT+M) management systems. Statistical significance between management systems of interest at three depths is depicted in the table insert.

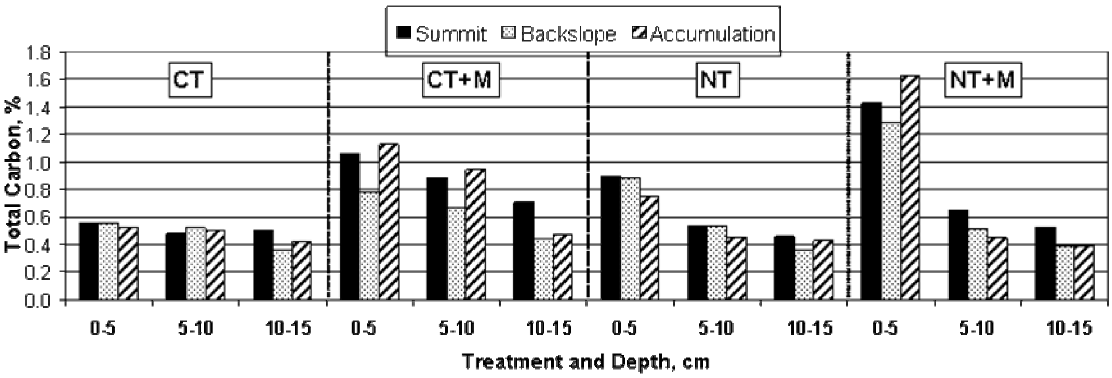

Figure 3. Total soil carbon content by landscape position, depth and management system (CT - conventional; CT+M - conventional with manure; NT - no-till; NT+M - no-till with manure)

Overall, non-inversion tillage increased infiltration in all zones. The NT system had greater infiltration in the summit and accumulation zones than in the backslope. A similar trend was noted with NT+M. The backslope position is a transitional zone where C deposition and accumulation is less likely to occur. Infiltration in the summit for the CT treatment was greater than in the accumulation and backslope zones. Manure application did not improve infiltration within tillage system in the study area, suggesting tillage had a greater influence on infiltration. No main effect for treatment (P = 0.51) and zone (P = 0.27) was observed for aggregate stability. This may be attributed to the large variability in aggregate stability measurements.

Conclusion

Manure significantly increased C content in CT and NT treatments, especially in the 0-5 cm of depth. However, it did not improve infiltration or aggregate stability. There were no significant differences between treatments and zones in aggregate stability. Infiltration tended to be higher in the summit position for all the treatments, with the exception of NT. Overall, conservation systems have improved C contents and infiltration of this landscape.

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Influence of feedstock and production conditions on biochar stability (short and long-term) and soil functional attributes

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Abstract

Biochar is receiving much attention as a potential tool for mitigating climate change through long-term biological carbon sequestration. However, before this potential can be recognised, it is essential that the production (usually pyrolysis) and sequestration processes are effectively designed and tested. It is important that biochar produced from different feedstocks under different processes is shown to have no detrimental effects on the environment. The beneficial agronomic benefits of biochar additions to soils, as well as short and long-term stability of biochar in soils also need to be demonstrated before biochar is widely implemented. If biochar is to be effectively used for carbon sequestration on a large-scale, its long-term stability (i.e. centuries to millennia) needs to be proven. It has also been shown that biochar contains a fraction of relatively labile carbon, which will affect the short-term stability of biochar in soil (as well as not being eligible for carbon credit under current schemes). Depending on the size of this labile fraction, optimisation of biochar production parameters to enable maximum carbon retention may therefore be counter-productive.

The main objective of this work has been to develop a toolkit which provides a means for rapid screening of different biochar products. The toolkit took the form of five different assays designed to test the stability of biochar in soil, the effect of biochar additions on pre-existing soil carbon, and the agronomic value of biochar products. Short-term stability (i.e. quantification of the labile carbon fraction of biochar) was assessed using controlled incubations of biochar in sterilised sand. Long-term stability was tested by subjecting biochar to a novel accelerated ageing technique. Any priming (and its magnitude) for the loss of pre-existing soil carbon was determined by incubating biochar in a range of different soils, using standard stable isotope techniques. The nutrient value of biochar was determined using a procedure to extract mineral ions determined to be ultimately crop available, whilst the soil structural value of biochar products was evaluated using an approach that assessed the addition of biochar to abiotic and biotic aggregation processes in soil.

The results of this work will be used to assess how biochar production conditions can be optimised, in order to produce biochar with properties that provide the most useful functions, relevant to the situation in which it is used.