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Effects of land-use intensification on soil carbon and ecosystem services in Brigalow (*Acacia harpophylla*) landscapes of southeast Queensland, Australia

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Abstract

Soil organic matter is an effective indicator of soil resource condition that reflects functional traits such as aggregation and infiltration and plays a critical role in sustaining production and ecosystem services in agricultural landscapes. Agricultural practices typically reduce soil carbon through the action of soil disturbance and mineralization. In the Brigalow (*Acacia harpophylla*) landscape we studied, soil carbon levels in pellic vertisols were significantly lower in the agricultural matrix of cropping and grasslands than in remnant Brigalow vegetation. There was no detectable gradient of soil carbon across Brigalow/matrix boundaries. Uncultivated grasslands showed significantly higher carbon levels than currently and previously cultivated grasslands. Regenerating grasslands showed no significant recovery of soil carbon over 15 years. Total, organic and labile soil carbon fractions were used to indicate different aspects of soil function, with the more active (labile) components reacting more sensitively to changes in land management. The carbon management index (CMI) was used to combine the active and passive components of soil carbon to provide a sensitive indicator of the rate of change of carbon dynamics in response to changes in land management at local-scales. A landscape CMI (CMI_L) was developed using a combination of soil samples and GIS-derived spatial data and is proposed as a potentially useful tool for modelling soil carbon dynamics in agro-ecosystems at local and landscape scales.

Keywords: Soil carbon; Ecosystem services; Landscape carbon management index

1. Introduction

The role of soil organic matter (SOM) in maintaining soil structure and productivity in agro-ecosystems is well recognised (Dalal et al., 1991; Carter, 1996; Karlen et al., 1997). SOM has been widely used as an effective indicator of the functional response of soils to land-use intensification (Carter, 1996; Whitbread et al., 1998; Dalal et al., 2003). For example, declines of soil organic matter with increasing farming intensity and duration have been well documented in Australia (Dalal and Mayer, 1986a; Gifford et al., 1992; Golchin et al., 1995) and overseas (Lucas et al., 1977; Mann, 1986; Spaccini et al., 2001; Lemenih et al., 2005). SOM decline usually results from changes to soil structure caused by tillage, removal of biomass and increased mineralization and decomposition of exposed soils (Oldeman et al., 1990).

Soil organic carbon (SOC) has been used in the majority of recent monitoring programs to assess soil resource condition and trends (e.g. Cornforth, 1999; Haynes, 2000; Sparling et al., 2004; Karlen et al., 2001; Farquharson et al., 2003) and is a useful surrogate measure that integrates the influence of many different land management factors. However, short-term changes in SOC in response to altered land management are usually difficult to detect (Blair et al., 1995). Labile carbon fractions are increasingly being used in agro-ecosystems research because they respond more sensitively to changes in land management such as tillage

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(Dalal and Mayer, 1986c; Biederbeck et al., 1994; Blair et al., 1995; Haynes, 2000). Labile fractions of carbon are closely linked to soil microbial biomass and organic matter turnover and can serve as indicators of the key chemical and physical properties of soils such as infiltration (Bell et al., 1998) and the availability of labile nutrients such as nitrogen, sulphur and phosphorus (Powlson et al., 1987).

To address these issues, Blair et al. (1995) combined labile and non-labile carbon fractions to derive a carbon management index (CMI). Compared with a single measure such as organic carbon, CMI can be used as a more sensitive indicator of the rate of change of soil organic matter in response to land management changes, relative to a more stable reference soil. To date, the CMI has been used to monitor carbon pools only at farm or paddock scales (e.g. Blair et al., 1995; Whitbread et al., 2003). We propose that CMI may also be potentially useful for measuring broader scale landscape function and soil ecosystem services.

In Australia, the conversion of extensive areas of native vegetation to pastoral and agricultural lands has lead to widespread degradation of soil structure and fertility of arable landscapes (Dalal and Mayer, 1986a; Commonwealth of Australia, 2002). For example, the Brigalow Belt Bioregion of southern Queensland has undergone recent, extensive land clearing and subsequent fragmentation of native vegetation. Between 1997 and 1999, the annual rate of vegetation clearing in Queensland was 4460 km^2 of which over 60% occurred in the Brigalow Belt Bioregion (Wilson et al., 2000). Under natural conditions, Brigalow (*Acacia harpophylla*) vegetation is usually associated with deep, fertile and well structured cracking clays (vertisols) on lowlands (Sattler and Williams, 1999). Consequently the majority of Brigalow lands have been converted to agricultural lands, with semi-natural grassland areas increasingly being converted to cropping. Such agricultural intensification often leads to local-scale changes in soil carbon levels (Dalal and Mayer, 1986b) and may have broader implications for landscape functioning (Matson et al., 1997), carbon sequestration (Falloon et al., 2002) and climate change (Paustian et al., 2000).

In this paper, we compare total, organic and labile soil carbon in soil samples from the core and edge of patches of remnant Brigalow vegetation with those from the core and edge of adjacent managed grasslands and cultivated land (the production matrix). We also investigate the management history of the matrix and its effects on selected soil carbon fractions. We use total and labile carbon fractions to derive a carbon management index (CMI) as developed by Blair et al. (1995). We propose a new 'landscape carbon management index' (CMI_L) as a tool for interpreting locally derived carbon measurements across different spatial scales. We discuss appropriate management options for maintaining soil function and consider our research findings in the context of ecosystem services.

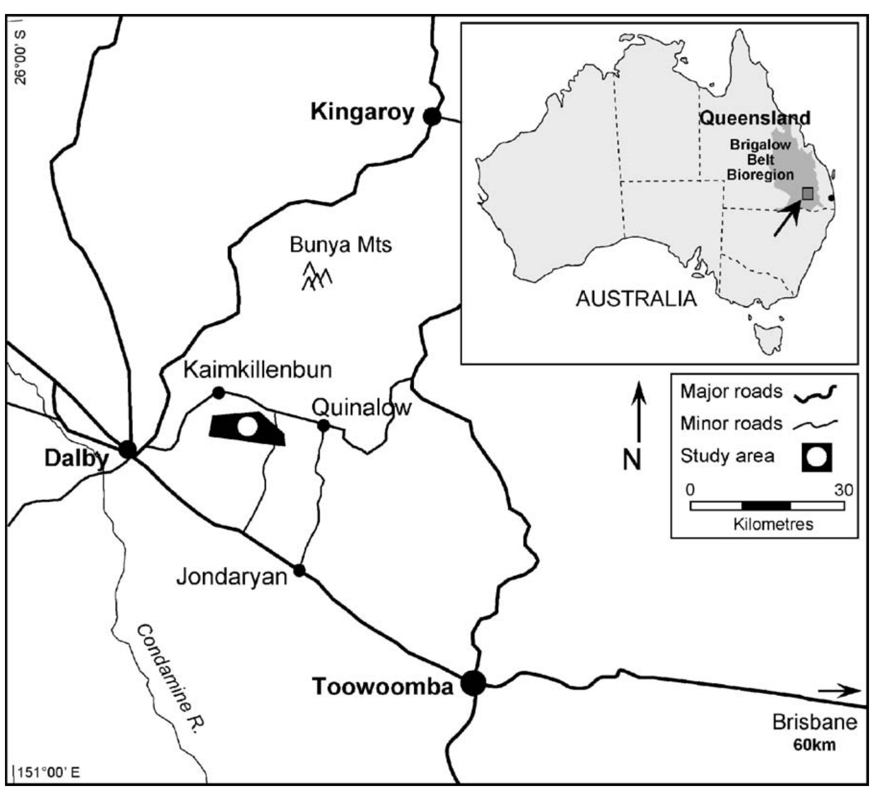

Fig. 1. Map of Queensland, Australia, showing the Brigalow Belt Bioregion and the location of the study area in southeastern Queensland.

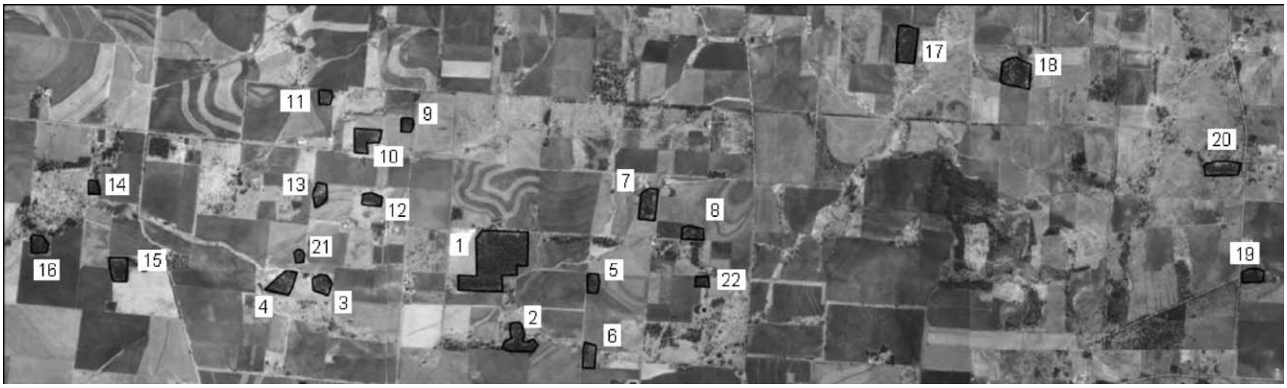

Fig. 2. Aerial photograph of the study landscape taken in 2001. Brigalow study sites are outlined and numbered.

2. Materials and methods

2.1. Site and soil description

The field research was conducted in the Eastern Darling Downs province of the southern Brigalow Belt Bioregion in Queensland, Australia. Nearby Dalby (lat. 27°18′S, long. 151°26′E), lies 340 m above sea level (Fig. 1).

The climate is sub-tropical with an average annual rainfall of 676 mm, mostly during the summer months (Bureau of Meteorology, 2005). Mixed farming systems with summer and winter cropping, interspersed with some grazing, are the predominant land-uses in the area. Brigalow soils are typically pellic vertisols (FAO/Unesco, 1974), characterised by high nitrogen content resulting from nitrogen fixation by root-nodulating bacteria (*Rhizobium* spp.) associated with Brigalow root systems (Isbell, 1962). Surface soil samples have pH 8–9 and are slowly permeable.

Unlike other areas of the Brigalow Belt Bioregion which have undergone recent broad-scale land clearing, the amount of original Brigalow vegetation in the study area has remained virtually unchanged at about 6% since the 1970s. After initial clearing of the pre-European native woody vegetation in the 1940s, often involving deep ripping and burning, some of the land was used for cropping. However, much of the cleared land regenerated, forming semi-natural grassland (Johnson, 1964). In 1945, only 30% of the landscape was used for cropping. Since 1945, much of this grassland has been gradually converted to intensively managed cropping lands and, today, only 25% of non-cropping land (i.e. grassland and Brigalow vegetation) remains in the landscape (Fig. 2).

2.2. Study design

Twenty-two study sites were selected within Brigalow (*A. harpophylla*) and Belah (*Casuarina cristata*) shrubby open forest (henceforth referred to as 'Brigalow') on cracking Cainozoic clay soils (defined by Sattler and Williams, 1999 as 'Regional Ecosystem' 11.9.5). Sites were selected from the mosaic of available remnant patches in the landscape. The size of Brigalow patches sampled ranged from 2 to 33 ha.

The study consisted of two comparisons:

(1) *Landscape comparison*: treatments were Brigalow remnant core (BC), Brigalow remnant edge (BE), production matrix core (MC) and production matrix edge (ME). Matrix treatments were adjacent to selected Brigalow remnant patches (Fig. 3).
(2) *Matrix comparison*: treatments were: (1) grassland that has been left uncultivated following clearing of the original Brigalow vegetation in the 1940s ($Grass_u$), (2) grassland that has been previously cultivated but left fallow for 3–15 years ($Grass_p$), and (3) recently (<2 years) cultivated land (Cultivation) (Fig. 3).

BC was located at the centroid of each of the 22 Brigalow patches and the other treatments (BE, ME and MC) were placed along a transect from BC (Fig. 3). The distance between ME and MC was either proportional to that between BC and BE, or a minimum of 100 m.

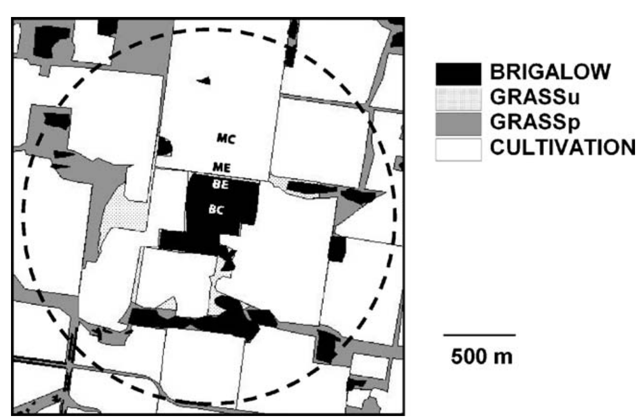

Fig. 3. Digitised schematic of a study site and the surrounding landscape showing: (1) landscape comparison treatments at Brigalow core (BC) and Brigalow edge (BE) and at the matrix core (MC) and matrix edge (ME), and (2) matrix comparison treatments of uncultivated grassland ($Grass_u$), previously cultivated grassland ($Grass_p$) and cultivation. The dashed circle represents a 2000 m buffer area around BC.

Table 1
Summary of the methods used for measuring total, organic and labile carbon fractions

Carbon fraction	Method	Reference
Total carbon	Weight loss-on ignition	Storer (1984)
Organic carbon	Wet combustion, Cr_2O_7 oxidation + colorimetric analysis	Walkley and Black (1934) and Sims and Haby (1971)
Labile carbon	Oxidation 333 mM $KMnO_4$ + spectral analysis (565 nm)	Blair et al. (1995)

Aerial photographs from 1945, 1971 and 2001 were scanned and registered using field-derived GPS co-ordinates. The area of Brigalow vegetation, grassland and cultivated land within a 2000 m buffer area around each study site was mapped and ground-truthed using ArcView version 3.2 (Environmental Systems Research Institute, 1996). A mean percentage cover estimate for the landscape was derived using percent cover values from each study site.

2.3. Sampling and soil carbon analysis

Soil cores (10 cm diameter) were taken to 7.5 cm depth from five sampling points, no less than 2 m apart within each treatment (i.e. BC, BE, ME and MC) at all 22 sites ($n = 88$). This sampling depth was chosen to reflect the higher surface layer responsiveness of carbon to changes in land management (Janzen et al., 1992; Biederbeck et al., 1994). Surface litter was removed prior to soil coring and samples from each treatment were pooled and thoroughly mixed.

Pooled soil samples were sieved (<5 mm) to remove coarse debris (e.g. roots), air dried, mechanically ground to the appropriate particle size and tested for total, organic and labile carbon concentrations using the methods described in Table 1. Bulk density measures were not used because previous CMI calculations are based on concentration, rather than area values (e.g. Blair et al., 1995; Conteh et al., 1998; Whitbread et al., 1998, 2003; Blair and Crocker, 2000).

2.4. Carbon management index

We calculated the carbon management index (CMI) using mean values for total and labile carbon from Brigalow core (BC) and matrix core (MC) measurements across all sites using the procedure proposed by Blair et al. (1995):

(1) carbon pool index (CPI) = total C_{Matrix}/total C_{Brigalow};
(2) lability index (LI) = lability of C_{Matrix}/lability of C_{Brigalow} where lability of $C = C_{\mathrm{Labile}}/C_{\mathrm{Non\text{-}Labile}}$;
(3) carbon management index (CMI) = CPI × LI × 100.

2.5. Landscape carbon management index

A 'landscape CMI' ($\mathrm{CMI_L}$) was derived by multiplying the CMI value for three land-uses: (1) Brigalow vegetation, (2) grassland (=$\mathrm{Grass_u}$ and $\mathrm{Grass_p}$ combined), and (3) cultivated land, by the proportion (%) of that land-use in the landscape (L_P). The resulting values for each land-use were then summed to give the $\mathrm{CMI_L}$:

$$\mathrm{CMI_L} = \sum_n (\mathrm{CMI} \times L_\mathrm{P})$$

2.6. Statistical analyses

One-way analysis of variance (ANOVA) was used to compare means of the three carbon fractions for all treatments using SPSS version 12.0.1 (SPSS Inc., 2003). Differences between treatments were analysed using a priori paired contrasts between all possible combinations of treatments.

Percentage data for total and organic carbon fractions were arcsine square-root transformed prior to analysis to overcome the problem of bounded proportion data. Residual plots were normally distributed, indicating that the assumptions of the ANOVA were appropriate for the data.

3. Results

3.1. Land-use change 1945–2001

Comparisons of landscape percentage cover showed that between 1945 and 2001 native woody vegetation cover had decreased across the landscape from approximately 17% to 6%, while grassland cover had decreased from 54% to 20%. During this time cropping had increased from 30% to 75%, largely a result of conversion of semi-native grasslands. Relatively small changes to grassland and cultivated cover have occurred since 1971 (Table 2).

3.2. Landscape comparison

Across all Brigalow and matrix treatments, total soil carbon ranged from 0.7% to 9.2%, organic carbon ranged

Table 2
Percent of study landscape under Brigalow, grassland and cultivation in 1945, 1971 and 2001

Year	Brigalow	Grassland ($\mathrm{Grass_u} + \mathrm{Grass_p}$)	Cultivation
1945	16.5 (1.7)	53.7 (0.7)	29.8 (1.9)
1971	5.8 (0.2)	26.9 (1.7)	67.4 (1.7)
2001	5.8 (0.3)	19.5 (0.6)	74.6 (0.5)

Values are means (±S.E.) derived using aerial photographs of the landscape around 22 study sites.

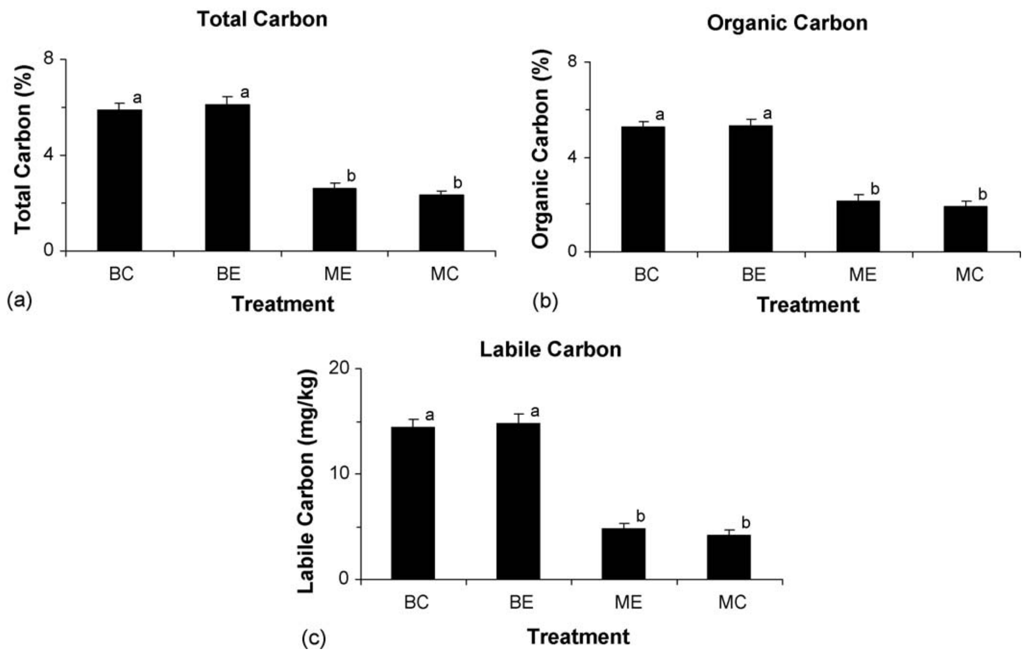

Fig. 4. Mean values for total (a), organic (b) and labile (c) carbon at Brigalow core (BC), Brigalow edge (BE), matrix edge (ME) and matrix core (MC); bars represent S.E. Means sharing the same letter are not significantly different (a priori contrasts $p > 0.001$).

from 0.5% to 8.3% and labile carbon ranged from 1.3 to 24.0 mg/kg. Mean values for carbon fractions in each treatment for the three carbon fractions are displayed in Fig. 4.

There were significant differences between landscape treatments for total carbon (Fig. 4a, ANOVA $F_{3,84} = 69.9$, $p < 0.001$), organic carbon (Fig. 4b, ANOVA $F_{3,84} = 65.3$, $p < 0.001$), and labile carbon (Fig. 4c, ANOVA $F_{3,84} = 73.4$, $p < 0.001$). However, there were no significant differences between edge and core measurements in the same land-use (i.e. Brigalow or matrix) for any of the three carbon fractions (Fig. 4).

3.3. Matrix comparison

There were no significant differences in carbon levels between matrix edge (ME) and matrix core (MC) treatments (a priori contrasts $p > 0.05$), allowing values to be pooled for the matrix comparison. Mean values for residual carbon, relative to reference (Brigalow) soils for each matrix treatment are shown in Table 3.

There were significant differences between matrix treatments for total carbon (Fig. 5a, ANOVA $F_{2,41} = 22.6$, $p < 0.001$), organic carbon (Fig. 5b, ANOVA $F_{2,41} = 28.0$, $p < 0.001$), labile carbon (Fig. 5c, ANOVA $F_{2,41} = 45.8$, $p < 0.001$) and carbon management index (CMI) (Fig. 5d, ANOVA $F_{2,19} = 14.3$, $p < 0.001$). Uncultivated grassland sites were significantly different from previously (and currently) cultivated sites for all three carbon fractions (a priori contrasts $p < 0.05$, Fig. 5). However, there was no significant difference between currently (Cult) and previously cultivated ($Grass_p$) treatments (Fig. 5).

3.4. Residual levels of soil carbon and CMI relative to Brigalow

The overall proportion of soil carbon at matrix core (MC) relative to soil at the Brigalow core (BC) was highest for total carbon and lower for organic and labile fractions; an exception was organic carbon in the uncultivated grasslands (Table 3). A higher proportion of residual carbon was found in uncultivated grassland soils compared with previously and currently cultivated soils (Table 3).

The CMI was highest in uncultivated grasslands and lowest in currently cultivated areas, although there was no significant difference ($p > 0.05$) between cultivated and previously cultivated areas (Fig. 5d). There were significant differences between the CMI in uncultivated grasslands and both previously and currently cultivated grasslands ($p < 0.05$) (Fig. 5d).

Table 3
Summary of residual carbon for the matrix comparison.

Matrix treatments	n	Residual carbon levels (%)		
		Total C	Organic C	Labile C
$Grass_u$	3	63.3 (8.6)	71.4 (11.6)	62.0 (13.2)
$Grass_p$	5	40.5 (4.2)	34.9 (6.1)	28.4 (5.6)
Cultivation	14	33.9 (2.8)	29.3 (2.5)	22.5 (1.8)

Residual levels are carbon values expressed as a percentage of soil carbon in reference soils under remnant vegetation. Values are means (±S.E.) for total, organic and labile carbon from pooled matrix data. '$Grass_u$' is uncultivated grassland and '$Grass_p$' is previously cultivated (regenerating) grassland.

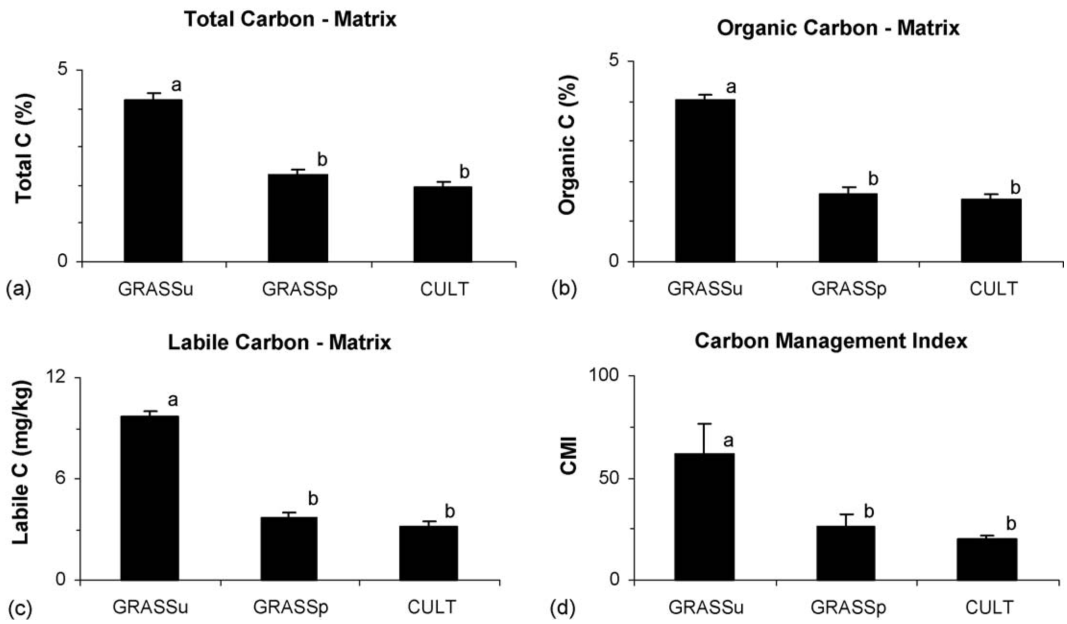

Fig. 5. Mean values for: (a) total, (b) organic, (c) labile carbon and (d) CMI in the matrix. $Grass_u$ = uncultivated grassland, $Grass_p$ = previously cultivated grassland and Cult = currently cultivated; bars represent S.E. Means sharing the same letter are not significantly different ($p > 0.05$).

3.5. The landscape CMI: application across land-uses

The study landscape comprised 6.3% Brigalow vegetation, 74.7% cultivated land and 18.9% grassland (includes previously and currently cultivated), and the CMI_L for this landscape is 29.0 (Table 4).

4. Discussion

4.1. Soil carbon decline in agro-ecosystems

The concentration and rate of decline of soil carbon vary considerably over time according to soil type, depth of measurement, rainfall, vegetation communities and management practices (Weil and Magdoff, 2004). For example, Dalal et al. (2005) found lower organic carbon levels in surface soils (0–10 cm) under mulga (0.69%) (*Acacia aneura*) than under Brigalow (2.23%) (*A. harpophylla*) soils. Dalal and Mayer (1986b) also recorded organic carbon levels of around 2% in uncultivated reference soils (0–10 cm) compared with an average of over 5% for our remnant Brigalow sites (0–7.5 cm). This difference may be caused by variation in the depth of sampling, higher levels of plant organic matter and charcoal, or by past clearing or grazing management practices. However, despite these differences, the relative changes in soil carbon levels in our study are consistent with those observed in surface soils under long-term cultivation in other studies (e.g. Dalal and Mayer, 1986b; Whitbread et al., 1998).

Similar to our findings, other Australian studies have also shown greater declines of labile carbon compared with total carbon, indicating higher sensitivity of the labile fractions to farm management practices (Blair et al., 1995). For example, Blair and Crocker (2000) recorded average losses of 56% and 71% of total and labile carbon, respectively, under a range of crop management practices in black soils (vertisols) near Tamworth, Australia after 20 years of cropping. Whitbread et al. (1998) recorded average losses of 37% total carbon and 54% labile carbon in grey clays (vertisols) after cropping durations ranging from 2 to 43 years (Table 5).

We consider the combination of the different carbon fractions used in the CMI a more informative indicator of overall change in carbon dynamics across the landscape than a single measure such as organic carbon. The CMI values for various Australian studies are compared in Table 5. To date, the majority of studies that have used the CMI have based their calculations on concentration values. However, future studies should consider compensating for differences in soil bulk density, by expressing carbon content on an area, rather than concentration basis. Problems associated with volumetric analyses and quantifying bulk density in cracking clays soils (e.g. vertisols), can be overcome by determining bulk density at drained upper limit (Dalgliesh et al., 1998).

Table 4
Proportion of the study landscape under each land-use (L_P), the carbon management index (CMI) for each land-use, the CMI_P ($=L_P \times$ CMI), and the landscape CMI ($CMI_L = \sum CMI_P$)

Land-use	L_P (%)	CMI	CMI_P
Brigalow	6.3 (0.3)	100	6.3
Cultivation	74.7 (0.6)	20.4 (1.7)	15.2
Grassland	18.9 (0.6)	39.5 (8.8)	7.5
$CMI_L(\sum CMI_P)$			29.0

Values for L_P are means of percent cover (±S.E.) within a 2000 m radius of 22 study sites.

Table 5
Decline of total and labile soil carbon, and the carbon management index (CMI), in various Australian surface soils (≤10 cm) under dryland cropping

Study (year)	Soil type	Cropping duration (years)	Total C (%)	Labile C (%)	CMI
Blair et al. (1995)	Red earth (paleustalf)	16–18	50.4 (4.8)	66.9 (7.5)	30.5 (7.5)
Blair et al. (1995)	Brown earth (palexeralf)	4	30.2 (0.0)	50.9 (0.0)	45.0 (0.0)
Blair et al. (1995)	Black earth (pellustert)	7	47.5 (0.0)	57.2 (0.0)	33.0 (0.0)
Whitbread et al. (1998)	Red earth (alfisol)	15–40	64.4 (4.4)	71.2 (3.5)	28.0 (4.0)
Whitbread et al. (1998)	Grey clay (vertisol)	2–43	36.8 (5.9)	54.0 (4.2)	44.0 (4.3)
Whitbread et al. (1998)	Black earth (vertisol)	25–40	37.3 (2.2)	33.6 (8.6)	66.5 (9.5)
Blair and Crocker (2000)	Black earth (pellic vertisol)	>20	56.0 (1.0)	70.6 (1.5)	26.7 (1.4)
Blair and Crocker (2000)	Red clay (chromic vertisol)	>20	45.8 (2.3)	46.6 (2.7)	53.3 (2.9)
Whitbread et al. (2000)	Red earth (ferric luvisol)	>18	74.2 (1.2)	76.4 (2.9)	23.0 (3.4)
Collard and Zammit (this study)	Grey-brown clays (vertisols)	>40	67.1 (2.8)	78.0 (1.9)	20.4 (1.7)

Values are means (±S.E.), expressed as a percentage of soil carbon in reference (uncultivated) sites, across land-use treatments in each study.

4.1.1. Application of the CMI at a landscape scale

Agricultural landscapes are traditionally managed at paddock or farm scales, with poorly understood links between production activities and landscape scale ecosystem processes. Similarly, the CMI has been effectively used in the past to compare short-term changes to carbon dynamics in cultivated areas and grasslands at the paddock or farm scale (Blair et al., 1995; Conteh et al., 1998; Whitbread et al., 1998, 2003; Blair and Crocker, 2000). Our research extends the application of the CMI to the landscape scale, by extrapolating local soil measures to broader scales using GIS-derived land-use classifications including woody vegetation, grasslands and cropping. The landscape carbon management index (CMI_L) can be used by land managers to combine the passive and active components of soil organic matter, and to enable decision-making across spatial scales.

Other models have been used to link field-based soil measurements (including carbon) with spatially explicit databases such as geographic information systems (GIS) by aggregation, to produce broad-scale maps of soil resources and their response to changing land-use patterns (e.g. Parton et al., 1987; Coleman and Jenkinson, 1996; Paustian et al., 1997). In particular, Park and Cousins (1995) proposed a method for integrating GIS data with locally derived soil health measures to provide management guidelines across scales. They identified the deficiencies of monitoring only the stable fractions of the soil organic matter pool and proposed the use of the more active biotic elements of soil. The CMI incorporates both the stable and responsive (biologically derived) fractions of the SOM pool, making it an ideal candidate for use in such models.

The significant differences in soil carbon between remnant Brigalow and the agricultural matrix were not apparent between core and edge in the same land-use (Fig. 3). This observed relationship across land-use boundaries suggests that, in our landscape, different land-uses (e.g. cultivated paddocks or patches of vegetation) can be considered discrete management units, in terms of soil carbon dynamics, based on their current soil carbon properties and management history. This defined boundary effect and observed homogeneity within land-uses enables accurate extrapolation of local measurements to larger scales using GIS-based methods.

The CMI_L has the potential to be useful across landscapes of varying size and composition, and locally derived measures of soil carbon can be accurately extrapolated in multi-use landscapes, provided that the types and duration of land management are well defined. In the landscape we studied, only three broad land-use categories (Brigalow vegetation, grassland and cultivation) were used to classify the landscape. Further division and spatial mapping of grassland and cropping types based on management regimes in the matrix (e.g. grazing, tillage practices, crop rotations) would provide additional information and allow stronger inference about the impact of land management on carbon pools across the landscape. Spectral imaging could be used to differentiate more accurately between land-uses (Wessman, 1991).

4.2. Response of SOM to land-use intensification

Our research shows a marked decline in total, organic and labile carbon and CMI as land-use shifts along the intensification gradient from remnant native vegetation (Brigalow) to cultivated land (see Table 2). Other studies have shown similar declines in carbon in surface soils in cultivated areas compared with virgin soils, as a result of exposure from tillage and subsequent mineralization of organic compounds (Blair and Crocker, 2000; Solomon et al., 2000; Murty et al., 2002; Dalal et al., 2005). This land-use intensification represents a large loss of soil carbon and a corresponding reduction in soil functional capacity across the landscape.

Contrary to our findings, other studies have detected higher carbon levels in uncultivated grasslands than in adjacent woody vegetation (e.g. Jenny, 1941; Rasmussen and Collins, 1991; Conant et al., 2001; Murty et al., 2002). In Brigalow systems, lower soil carbon levels in uncultivated grasslands may be due to soil exposure and mineralization caused by disturbances associated with clearing original woody vegetation and managing persistent regrowth (Johnson, 1964).

4.3. Management options for enhancing soil carbon

Soil carbon levels showed a limited capacity to recover in previously cultivated areas following regeneration

(<15 years) by semi-natural grassland communities. A similar inability of soil carbon levels to recover after 14 years was found in successional old-fields in Virginia by Riedel and Epstein (2005). However, in contrast to these findings, other studies have shown that regenerating grasslands following cultivation can significantly enhance soil carbon levels (e.g. Haynes, 2000; Conant et al., 2001; San Jose et al., 2003; Al-Kaisi et al., 2005). Total recovery of carbon to pre-agricultural levels after abandonment in a 'midcontinental' climate may take as long as 230 years (Knops and Tilman, 2000).

Possible reasons for the slow recovery of carbon in our study include drier than normal conditions in the landscape during the sampling period (Bureau of Meteorology, 2005), leading to low biological turnover (Six et al., 2002). Other factors include the short duration of regeneration (less than 15 years), the small number of grassland sites sampled, and uncertain records of grazing intensity and management practices at the time of initial Brigalow clearing.

Numerous studies have shown that SOM levels in agricultural lands can also be manipulated by changes to crop management such as crop rotation, tillage regimes, strategic fertiliser application and crop residue inputs (e.g. Grace et al., 1994; Haynes, 2000; Gregorich et al., 2001; Farquharson et al., 2003; Magdoff and Weil, 2004). Improvements in land management, such as crop rotations, have also been shown to increase labile fractions of soil carbon (Biederbeck et al., 1994; Bell et al., 1998; Whitbread et al., 2000).

4.4. Sustainable carbon management

Managing the chemical and structural properties of soils in agro-ecosystems is a vital component of sustainable production. In particular, soil processes such as nutrient cycling and carbon sequestration play an important role in regulating and maintaining beneficial ecosystem services (Daily et al., 1997). In this study, average soil organic carbon concentrations for cultivated land and previously cultivated grasslands were below the 2% considered to be the critical level of organic carbon required for maintenance of satisfactory soil aggregate stability (Greenland et al., 1975) and above which no further increases in productivity are achieved (Janzen et al., 1992). However, the quantitative basis for such thresholds, particularly in different soils, is limited (Loveland and Webb, 2003). Hence, in order to strengthen the applicability and interpretation of the CMI_L, research efforts should focus on refining threshold levels of total and labile carbon fractions from a range of soil types, below which soil functional capacity and the provision of ecosystem services diminish.

5. Conclusions

Soil carbon levels are widely used as an indicator of soil condition and may also reflect the duration and intensity of past agricultural management practices. In this study, total, organic and labile carbon concentrations were significantly higher in remnant Brigalow vegetation than in the adjacent agricultural matrix of cropping and grassland. In the matrix, uncultivated grasslands had significantly higher carbon levels than both currently cultivated areas and previously cultivated grasslands, with the latter showing no significant recovery of soil carbon after 3–15 years of natural grassland regeneration. Measurements of carbon concentration at the core and edge in the same land-use were not significantly different, indicating that there was no detectable gradient of soil carbon across land-use boundaries.

Consistent with other studies, active (labile) soil carbon components reacted more sensitively to changes in land management than total or organic carbon fractions. Hence, total and labile fractions were combined to derive the carbon management index (CMI), a more complete indicator of soil carbon dynamics. A new landscape CMI (CMI_L) was developed, using a combination of locally derived soil carbon measures and GIS-derived spatial data. The CMI_L is potentially useful as a spatially explicit tool for modelling the response of soil carbon and associated ecosystem services to changes in land management through time.

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