CSIRO PUBLISHING
www.publish.csiro.au/journals/ajsr *Australian Journal of Soil Research*, 2009, **47**, 273–285

# Accumulation of soil carbon under zero tillage cropping and perennial vegetation on the Liverpool Plains, eastern Australia

*R. R. Young*[A,D], *B. Wilson*[B,C], *S. Harden*[A], *and A. Bernardi*[A]

[A]NSW Department of Primary Industries, Tamworth Agricultural Institute, 4 Marsden Park Road, Calala, NSW 2340, Australia.
[B]NSW Department Environment and Climate Change, PO Box U245, University of New England, Armidale, NSW 2351, Australia.
[C]School of Environmental and Rural Sciences, University of New England, Armidale, NSW 2351, Australia.
[D]Corresponding author. Email: rick.young@dpi.nsw.gov.au

**Abstract.** Australian agriculture contributes an estimated 16% of all national greenhouse gas emissions, and considerable attention is now focused on management approaches that reduce net emissions. One area of potential is the modification of cropping practices to increase soil carbon storage.

Here, we report short–medium term changes in soil carbon under zero tillage cropping systems and perennial vegetation, both in a replicated field experiment and on nearby farmers' paddocks, on carbon-depleted Black Vertosols in the upper Liverpool Plains catchment.

Soil organic carbon stocks ($C_S$) remained unchanged under both zero tillage long fallow wheat–sorghum rotations and zero tillage continuous winter cereal in a replicated field experiment from 1994 to 2000. There was some evidence of accumulation of $C_S$ under intensive (>1 crop/year) zero tillage response cropping. There was significant accumulation of $C_S$ (~0.35 Mg/ha.year) under 3 types of perennial pasture, despite removal of aerial biomass with each harvest. Significant accumulation was detected in the 0–0.1, 0.1–0.2, and 0.2–0.4 m depth increments under lucerne and the top 2 increments under mixed pastures of lucerne and phalaris and of C3 and C4 perennial grasses. Average annual rainfall for the period of observations was 772 mm, greater than the 40-year average of 680 mm. A comparison of major attributes of cropping systems and perennial pastures showed no association between aerial biomass production and accumulation rates of $C_S$ but a positive correlation between the residence times of established plants and accumulation rates of $C_S$. $C_S$ also remained unchanged (1998/2000–07) under zero tillage cropping on nearby farms, irrespective of paddock history before 1998/2000 (zero tillage cropping, traditional cropping, or ~10 years of sown perennial pasture).

These results are consistent with previous work in Queensland and central western New South Wales suggesting that the climate (warm, semi-arid temperate, semi-arid subtropical) of much of the inland cropping country in eastern Australia is not conducive to accumulation of soil carbon under continuous cropping, although they do suggest that $C_S$ may accumulate under several years of healthy perennial pastures in rotation with zero tillage cropping.

## Introduction

The Fourth Assessment Report of the Intergovernmental Panel on Climate Change (IPCC 2007) states unequivocally that the earth's climate is changing and that this change is strongly associated with human activity. Because of concern regarding greenhouse gas (GHG) emissions, considerable effort is now being focused both nationally and internationally on strategies that reduce GHG emissions or sequester additional atmospheric carbon. Total GHG emissions in Australia were estimated to be 576 Mt $CO_2e$ (equivalents) in 2006 (Department of Climate Change 2008), to which agriculture contributed ~90.1 Mt $CO_2e$ (16%). Therefore, considerable scientific attention has been focused on this sector to identify management approaches that reduce net emissions. One area of potential is the modification of cropping practices to increase soil carbon storage.

Of all soil constituents, organic matter suffers most from cultivation of cropping land in Australia (Dalal and Mayer 1986). Reports from the USA (Lal *et al*. 2003; Johnson *et al*. 2005; Franzluebbers 2005), South America (Diekow *et al.* 2005; Bayer *et al*. 2006; Zanatta *et al*. 2007), and Europe (Smith *et al*. 1998) show that adoption of zero tillage (which includes stubble retention usually in combination with other practices such as fertiliser addition and sometimes the growing of cover crops) in annual cropping systems has the potential to accumulate total soil carbon stocks ($C_S$) at rates of up to 400 kg/ha.year. However, there is little evidence for this in the Australian literature (Dalal and Chan 2001; Chan *et al*. 2003;

 10.1071/SR08104 0004-9573/09/030273

Wang *et al.* 2004). Most Australian work in this area suggests that a phase of perennial pasture is needed to make net additions to $C_S$ (Holford 1981; Dalal *et al.* 1995; Young *et al.* 2005) while improved management of cropping systems only reduces the rate of decline in $C_S$ (Dalal *et al.* 1995; Dalal and Chan 2001; Heenan *et al.* 2004; Wilson *et al.* 2008).

Although differences in carbon levels between fallow management practices on Vertosols have been reported in favour of zero tillage, these differences have not been large and were dependent on the application of zero tillage, stubble retention, and nitrogen fertilisers (Dalal 1989; Dalal *et al.* 1995; Wang *et al.* 2004). On a coarse-textured soil in semi-arid central western New South Wales (NSW), of all management combinations, only the addition of fertiliser nitrogen to continuous wheat over 15 years resulted in a small (+0.03%) but significant increase in $C_S$ compared with the control treatment (Fettell and Gill 1995). However, serial measurements on a carbon-depleted Vertosol in south-eastern Queensland over 6 years showed no significant trends either up or down in $C_S$ under either conventional or zero tillage (Dalal *et al.* 1995). Although replacement of native woodland with traditional cultivation and annual cropping has invariably led to large declines in soil carbon (Dalal and Mayer 1986), replacement with perennial pastures does not guarantee an accumulation to pre-clearing soil carbon levels (Young *et al.* 2005).

Where comparisons have been made on carbon-rich soils, the rate of soil carbon depletion under reduced or zero tillage was typically found to be significantly lower than that under conventional tillage systems (e.g. Heenan *et al.* 2004, 1995). Conversely, we would expect that the rate of accumulation of $C_S$ would be greater in carbon-depleted soils than in carbon-rich soils and that the greatest efficiency in soil carbon sequestration will be in soils furthest from carbon saturation or equilibrium levels (Stewart *et al.* 2007).

Reports relating to soil carbon typically have measures over 1 or 2 years and, usually, some time after the application of tillage treatments (Dalal 1989; Fettell and Gill 1995; Wang *et al.* 2004). Although informative, such comparisons between management practices reflect only the differences in $C_S$ at the time of sampling; without equally precise sampling over time from when treatments were first applied, we cannot deduce rates of loss or accumulation of $C_S$. Only if many decades of traditional cropping precede the comparison might it be reasonable to assume that the traditional or conventional treatment is at equilibrium and has remained unchanged (Follett 2001).

Australian assessments of carbon flux in annual cropping systems, whether in Mediterranean, temperate, or subtropical regions, have centred almost exclusively on winter cropping. Research (Ringrose-Voase *et al.* 2003; Paydar *et al.* 2005) on the Liverpool Plains has demonstrated that the higher cropping frequencies characteristic of zero-tillage response cropping mimics the hydrologic stability of perennial systems and reduces deep drainage below the root-zone to small amounts.

Here, we test the supposition that the higher cropping frequency and productivity of zero-tillage response cropping systems, where both summer and winter crops are grown depending on plant-available soil water, may yield positive fluxes in soil carbon. We report a longitudinal comparison of the changes in soil carbon under several zero-tillage cropping systems and perennial pastures in a medium-term (August 1994–May 2000) replicated field experiment (described by Ringrose-Voase *et al.* 2003 and Paydar *et al.* 2005). In addition, nearby farmers' paddocks, originally sampled in 1998–2000 (Young *et al.* 2005), where continuous zero tillage cropping was ongoing and traditional cropping practices were replaced with zero tillage and where perennial pastures were replaced with zero tillage cropping, were resampled in 2007 to provide $C_S$ data to verify the results of the field experiment.

## Methods

### Cropping and pasture systems experiment

The cropping and pasture systems experiment was established in August 1994 on the farming property 'Hudson' located in the foothills of the Liverpool Ranges (31.75°S, 150.45°E; average annual rainfall 684 mm with some summer dominance, average annual pan evaporation 1718 mm) (Ringrose-Voase *et al.* 2003). The experiment spanned 2 interbank areas in the middle of a hillslope of 3–4% overlying Miocene basalt (a Lever Gully soil landscape; Banks 1998). The surface soil was self-mulching and the profile to around 1 m depth was colluvial material with 75–80% clay, of which 90% was smectite. This was overlying >5 m of brown clay and variable amounts of caliche. The profile was classified as an Endocalcareous, self- mulching, Black Vertosol, non-gravelly, very fine/very fine, giant (Isbell 1996). These soils have characteristically high plant-available water-holding capacities of ~250 mm for annual crops, which, for crops planted into a soil profile at close to field capacity, provides a significant buffer against the erratic occurrence of rainfall. Before cultivation, the hillslope had been perennial grassland for >50 years, with a total soil carbon concentration ($C_C$) of ~30–40 g/kg (0–0.1 m), determined from an adjacent grassed waterway (there is virtually no carbonate carbon present in the surface layers of soils on the mid-slopes of these basaltic landscapes; Young *et al.* 2005). After 22 years of continuous cultivation and cropping, the site was lower in $C_C$ (~15 g/kg). Apart from nutrient deficiencies (N, P, S, Zn) corrected before and early in the experiment, there were no further edaphic constraints apparent (including salinity and sodicity) to plant growth.

### Experimental design

The experimental design consisted of 9 treatments, which differed in the type and residence times of vegetation (residence time (days) was the period of time when established, live plants were present):

Three long fallow (LF) rotations of spring wheat (*Triticum aestivum*) and grain sorghum (*Sorghum bicolor*) (~25% crop residence time). The 3 long fallow rotations (LF1, LF2, LF3) were set up so that wheat and sorghum were planted each winter and summer respectively.

Continuous, or short fallow, winter cereal (*T. aestivum–Hordeum vulgare*) (40% residence time, W).

Two opportunity, or response, cropping treatments (50–70% residence time): winter cereal (*T. aestivum–H. vulgare*)–mung bean

(*Vigna radiata*) (RC1); grain sorghum (*S. bicolor*)–winter pulse (chick pea, *Cicer arietinum*, in 1995 and 1998, field pea, *Pisum sativum*, in 1996) (RC2). Both were planted using a planting rule of 0.5 m of wet soil measured with a push probe. The RC2 treatment was slowly transformed into continuous sorghum after partial failure of all but the first chickpea crop due to disease.

Three perennial pasture treatments (~90% residence time): lucerne (*Medicago sativa* cv. Aurora) (P1); lucerne and phalaris (*Phalaris aquatica* cv. Sirolan) mixture (P2); Bambatsi panic (*Panicum coloratum* var. *makarikariense* cv. Bambatsi), Wallaby grass (*Austrodanthonia linkii* cv. Bunderra), and Queensland bluegrass (*Dichanthium sericeum* local ecotype) mixture (P3).

Treatment plots were 40 by 16 m, arranged in 4 replicate blocks of 9, two in each interbank area.

### Crop management

All crops were planted using zero tillage machines. Crop residues were retained and weeds were controlled with chemicals or removed by hand. Zinc was applied (15 kg Zn/ha) as zinc sulfate heptahydrate solution in August 1994. Phosphorus (10 kg P/ha) was applied to all crops at planting, as Triphos (1.4% S) in 1995 and 1996, but after sulfur (S) deficiency was observed in mungbean, as single superphosphate (9% P, 11% S) in the remaining years. Nitrogen (N, as urea) was applied between planting rows to grass crops at sowing. LF sorghum received 80 kg/ha, LF and short fallow (W) winter cereal 100 kg/ha, and RC2 sorghum and RC1 winter cereal 60–80 kg/ha. All perennial pastures received an annual topdressing of single superphosphate at 250 kg/ha. The perennial grasses were also topdressed with N as ammonium nitrate at 50 kg/ha.year.

Pasture biomass was measured 3–5 times each year (quadrat harvests) followed by cutting and removing all but 200–500 kg/ha of the aerial biomass.

### Plant growth and water use

Plant growth and water use of all systems were measured intensively and are described elsewhere (Ringrose-Voase *et al*. 2003). Briefly, all winter and summer crops with at least 1 season of fallow (long and short fallow) were planted at, or close to, the times recommended for the chosen varieties. The 0.5 m of wet soil planting rule dictated that crops in response cropping treatments were planted in all seasons except the dry autumn–winter of 1997. Sorghum–chickpea response cropping was marred by the failure of all but the first chickpea crop due to viral disease.

### Soil sampling, total carbon, and bulk density

Soil from each plot was sampled biannually. Two separate 43-mm-diameter cores, located using a predetermined grid, were taken and cut into 0–0.1, 0.1–0.2, 0.2–0.4 . . . 2.8–3.0 m increments. In addition, 20 randomly sampled, 20-mm-diameter surface cores of 0.1 m depth were taken and bulked within each plot. Samples were dried (forced draught oven dry at 40°C, ≥72 h) on the day of sampling, ground to <2 mm, and stored in 120-mL specimen containers.

Total carbon concentration ($C_C$, g/kg) was determined by Dumas combustion where samples were oxidised in a vario MAX CN elemental analyser (Elementar Analysensysteme GmbH, Germany) with controlled oxygen supply at high temperatures (~900°C) using copper oxide and platinised catalyst (in-house method 630, issued 1 December 2006, NSW Department of Primary Industries, Environmental Laboratory, Wollongbar NSW).

Some bias in C results from individual core samples was apparent and appeared to be associated with the order of processing in the laboratory; this effect was exacerbated by the low values of $C_C$ (~14 g/kg) relative to the Laboratory's routine LOR of 0.2%. Although this variation was within the LOR and would have been acceptable for most other purposes, it had some effect on regressions of $C_C$ over time in this dataset. Therefore, $C_C$ data from the depth increments 0–0.1, 0.1–0.2, 0.2–0.4 m were analysed after adjustment of each value using the concentration measured at 0.4–0.6 m depth for that core sample (samples from individual cores were processed consecutively in the laboratory). The value for each of the shallower depths for each core was adjusted by dividing by the 0.4–0.6 m value for that core and then multiplying by the mean of all 0.4–0.6 m values. As far as we could ascertain, there were no differences between treatments $C_C$ at this depth.

### Data analysis

The bulk density ($\rho_b$, Mg/m$^3$) values used to calculate carbon stocks ($C_S$, Mg/ha) in the Hudson experiment were those of a reference profile (Ellert and Bettany 1995; Wang *et al*. 2004) at the drained upper limit (DUL), determined on the site (Ringrose-Voase *et al*. 2003). These were 1.00, 1.13, 1.11, and 1.12 Mg/m$^3$ for 0–0.1, 0.1–0.2, 0.2–0.4, 0.4–0.6 m, respectively(the slightly larger value at 0.1–0.2 m was probably due to compaction after many years of traditional farming). These Vertosols have marked shrink/swell characteristics due to the high content (~70%) of smectite clay minerals (Kirby *et al*. 2003). Consequently, $\rho_b$ changes with moisture content ($\theta_g$), which can be different between sampling times and experimental treatments. (On the Hudson experiment there was no evidence of changes in $\rho_b$ due to treatment effects that were independent of $\theta_g$). The size of the error is such that if a Vertosol were sampled when dried to the crop lower limit (CLL where $\rho_b = 1.21$) and soon after when almost wet ($\theta_g = 45\%$, $\rho_b = 1.14$), and $C_S$ was estimated simply by the product of $C_C$ and $\rho_b$, the difference in the estimated $C_S$ values for the soil when dry and when wet would be ~5%. The use of a reference profile with a constant soil mass in each layer avoids this error in the estimations of $C_S$.

In addition, when estimating $C_C$ in Vertosols of different water contents at fixed depth increments (say 0–0.05 or 0–0.1 m, etc.), a small error is introduced when samples are taken from a dry soil compared with those taken from the same soil when it is wet. This occurs because a greater mass of soil material is sampled from each fixed depth increment of the dry soil, which has shrunk vertically (and horizontally) during drying, compared with the same soil when wet. For $C_C$, which is usually larger in any soil layer than in the layer below, fixed depth sampling from a shrunken dry soil, compared with a swollen wet soil, will result in dilution of what was (for the wet and swollen soil) a carbon-rich surface layer, with soil from the layer below. The $\rho_b$ of soil dried to near the CLL is ~6% greater than that of a wet profile (bulk density calculations from Gardner 1988). The samples from the Hudson experiment were, by necessity, taken when the surface layers were, to some extent, dry and ranged in $\theta_g$ by up to 12% between sampling times and experimental treatments. Over this range, $\rho_b$ changed by ~0.07 resulting in shrinkage of a 100-mm soil layer by ~5 mm. Compared with a 0–0.1 m sample taken when $\theta_g$ was, for example, 45%, a fixed 0–0.1 m sample taken when $\theta_g$ is at CLL (33%) will contain 5 mm from the next layer down. Compared with the same sample with $\theta_g$ of 45%, $C_C$ will have become diluted in proportion to the rate of decline in $C_C$ with depth. At 0.1 m depth, the greatest rate of change in C concentration ($C_C$) with depth, which was that found under perennial pastures, was ~0.06 g/kg less for each mm of depth just below 0.1 m. That is, a dry, compared with a wet, soil surface 0–0.1 m will have $C_C$ reduced by 0.9%. This error was deemed small, and to reduce complexity, was ignored.

The carbon data were analysed using the statistical software package ASReml (Gilmour *et al.* 2006), which fits linear mixed models by Residual Maximum Likelihood. The cubic smoothing spline approach of Verbyla *et al.* (1999) was used to model the changes in carbon over time. This approach partitions the response into 2 components, a linear trend and a smooth non-linear trend about the linear component. Treatment (cropping system, pasture type), time (days), and their interaction are fixed terms in the model allowing the prediction of intercepts and slopes for each treatment. A spline term fitted as a random term in the model estimates the overall smooth linear trend and a treatment spline interaction allows the estimation of a smooth non-linear trend for each treatment. A random term to account for replicate effects at each sampling time, and random terms to account for variation due to plot differences, were also included. The significance level for all tests was $P = 0.05$.

### *Resampling of farm paddocks*

The Yarramanbah Creek and Big Jacks Creek sites (Young *et al.* 2005), sampled in March 1998 and January 2000, respectively, were resampled in May 2007. These were located on Black Vertosols (Isbell 1996) situated within gently inclined (<2% slope) alluvial fans of the Windy Creek landscape grouping (Banks 1998). On both sites, nearby paddocks and cropping strips were sampled. These had a history of (*a*) >30 years of continuous cropping with zero tillage, practised at the former from 2000 and at the latter from around 1992; (*b*) ~5 years of traditional cropping followed by ~10 years of lucerne-based pasture which was replaced by zero tillage cropping in 2000 that continued until sampling in 2007; (*c*) remnant native vegetation (grassy woodland) that was intermittently grazed.

From 2000 to 2007, wheat, barley, pulses, and grain sorghum were grown at both sites. Cropping frequency was ~1 crop/year, nitrogen applications to winter cereals and grain sorghum were 100–115 kg/ha.year, and average aerial biomass yields were estimated to have been 8–11 t/ha.year (see Table 3).

In all, 175 cores were taken along the original transects, with 8–21 taken from each land-use at each site. Samples to 0.6 m depth were taken using a 1-m-long, 100-mm-diameter steel tube fitted with a hardened cutting tip of 94 mm diameter pushed into the ground by a tractor mounted, hydraulically operated, coring machine. The location of each core was selected at random, but not in permanent wheel tracks (‘tramlines’). Cores were taken successfully, irrespective of the presence or absence of cracks in the soil. Both live and dead herbaceous plants were cut level with the soil surface beforehand. Core contents were divided into dead unattached plant material (litter) and soil depth increments of 0–0.05, 0.05–0.1, 0.1–0.2, 0.2–0.4, 0.4–0.6 m. Below the surface, the sample included all plant crowns and roots except tree roots >2 mm, which were accounted for when assessing above ground tree biomass (Young *et al.* 2005). In self-mulching soils, the soil surface is blurred with sometimes quite recent, fresh dead plant material mixed into the surface layers down to ~0.05 m. We separated such material (>5 mm long that had been incorporated into the surface 0–0.05 m by rain and the self-mulching action of the soil) from the soil and placed it in the litter sample. This separation required particular care in header trails on cropped land and under trees where fine plant residues had become incorporated into the surface layers. Bulk density ($\rho_b$) at field soil water content was calculated directly from core dimensions, core wet and dry (40°C) weights, and subsample oven dry (105°C) moisture content.

### *Data analysis*

As the original samples taken in 1998 and 2000 were in 0.2-m increments, $C_C$ and $C_S$ comparisons between sampling times were best made on 0.2-m increments. Each 0–0.2 m $C_C$ value for the 2007 samples was calculated from the values of the smaller increments within that depth range appropriately weighted for depth and bulk density ($\rho_b$), such that it would be equivalent to a single bulked 0–0.2 m sample, comparable to the 1998–2000 samples.

For comparison between sampling times, $C_S$ data were calculated from $C_C$ data using $\rho_b$ from a reference profile at DUL. The reference profile was determined from the 2007 sample $\theta_g$ and $\rho_b$ data. Regressions of $\rho_b/\theta_g$ were used to estimate $\rho_b$ at DUL for each depth increment. The water content at DUL for each depth layer was assumed to be that determined nearby on Hudson. As these field determinations of $\rho_b$ of Vertosols are dependent on the prevailing $\theta_g$ determined when samples were taken, the statistics reported here apply only to these $\rho_b$ values measured at their prevailing $\theta_g$ for a particular site and depth. The $\rho_b$ comparisons are therefore only approximate. The changes in $\rho_b$ values that might be encountered should soils have been wetter or dryer by, for example, $\pm 0.1$ g/g are theoretically (Gardner 1988) ~($\pm 0.05$) Mg/m$^3$, i.e. a slope of –0.5; i.e., each $\rho_b$ value may vary within a range of ~0.1 Mg/m$^3$ depending on the prevailing $\theta_g$.

The $\rho_b$ at DUL in the 0–0.05 m layer measured under native vegetation was smaller (0.69 Mg/m$^3$) than that under cropping (0.80 Mg/m$^3$) and was not affected by water content (slope $\approx 0$), whereas $\rho_b$ under cropping was affected (slopes $\approx -0.5$). Therefore, this consistent difference in $\rho_b$ between cropped and never cropped land was incorporated into the calculation of the top layer of the reference profiles to avoid overestimation of $C_S$ under native vegetation. The $\rho_b$ values for the reference profiles were 0–0.2 m, 1.003 (cropping) and 0.977 (native vegetation); 0.2–0.4 m, 1.121; 0.4–0.6 m, 1.144.

Regression models were fitted to the carbon and bulk density data using the ASReml command in the ASReml-R software package (Butler *et al*. 2007). The $\rho_b$ data were analysed as univariate data using a repeated-measures approach. The fixed terms in the model were depth, site, land-use, and all interactions with field moisture content included as a covariate. Correlations between depths were modelled with a heterogeneous power structure. The $C_C$ data from the 2007 re-sampling were analysed as multivariate data with each depth taken to be a variable. These depth variables were correlated and the correlations were modelled with a series of covariance structures including uniform, power, antedependence, and unstructured, with an antedependence structure of order 1 being the most parsimonious. The fixed effects in the model were depth trait, site, land use, and all interactions. The comparison of $C_S$ data from the 1998–2000 initial sampling and the 2007 re-sampling were analysed in a similar way with the inclusion of date of sampling as a fixed effect.

## Results

### *Cropping and pasture systems experiment*

The weather from 1994 to 2000 ranged from drought in 1994 to an extremely wet winter in 1998. Summer rainfall exceeded winter rainfall in 4 of 6 years, with the 1995–96 summer the wettest (Table 1). Average annual rainfall was 772 mm over the duration of the experiment, 12% greater than the long-term (40 years) average.

Crops planted after long or short fallow produced the most biomass and grain per crop but the sorghum–chickpea system produced the greatest average annual biomass. Perennial pastures consistently produced less biomass than annual crops except mungbean and chickpea (Table 1).

Average baseline $C_S$ determined from the bulked 0–0.1 m samples collected in August 1994 was 13.92 Mg/ha and values determined from individual cores sampled at that time were 13.74 (0–0.1 m), 12.64 (0.1–0.2 m), and 24.13 (0.2–0.4 m) Mg/ha using the reference profile bulk density ($\rho_b$) values. There were no significant rates of change in total soil $C_S$ over time in the long fallow or continuous winter cereal cropping systems (Figs 1, 2) (Table 2). There was some evidence of increased $C_S$ under winter cereal–mung bean response cropping in the bulked (0–0.1 m) samples and in the 0.1–0.2 m core layer under sorghum–winter pulse response cropping. These isolated trends in $C_S$ accumulation under intensive cropping were in contrast to the significant, consistent increases in $C_S$ under all 3 perennial pastures, down to 0.2–0.4 m under lucerne (Fig. 3, Table 2), even though aerial biomass had been removed with each harvest.

The bulk sample $C_C$ tended to be larger than core 0–0.1 m values from continuous winter cereal and winter cereal–mung bean response cropping especially. We have no explanation for this, except for the possibility that winter cereal residues were more carefully removed from core samples that were laid out on a bench compared with the bulked samples, which were inspected briefly for large pieces of residue.

Accumulated biomass yields (Table 1) from perennial pastures were similar, but less than all cropping systems. Winter cereal–mungbean response cropping, although showing evidence of $C_S$ accumulation in bulked 0–0.1 m samples but not the core samples, produced the lowest biomass of all the cropping systems. Despite having the largest aerial biomass productivity of all systems, the sorghum–chickpea response cropping treatment registered significant accumulation of $C_S$ accumulation only at the 0.1–0.2 m depth increment.

**Table 1. Rainfall and seasonal aerial biomass yields of zero tillage cropping systems and perennial pastures on the Hudson field experiment**
Crop growing seasons are shown as summer (S) and winter (W); crops or pastures were not grown in a fallow season (f); 30–50% of crop biomass was removed as grain except where crops failed due to drought in 1994 and viral diseases of winter pulses (biomass yields are italicised). Pasture biomass was removed from the plots at each harvest leaving 0.2–0.5 Mg/ha of stubble. Total residence time (days) was the total of all periods when established, live plants were present

| System | 1994 | | 1995 | | 1996 | | 1997 | | 1998 | | 1999 | | Total | Av. aerial |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | W | S | W | S | W | S | W | S | W | S | W | S | residence | biomass |
| Rainfall (mm): | 112 | 269 | 243 | 538 | 360 | 369 | 236 | 329 | 708 | 433 | 644 | 394 | time (days) | (Mg/ha.year) |
| Long fallow wheat–sorghum rotations | | | | | | | | | | | | | | |
| Phase 1 | *0.5* | f | f | 13.2 | f | f | 11.7 | f | f | 13.3 | f | f | 470 | 6.45 |
| Phase 2 | f | 10.5 | f | f | 12.7 | f | f | 11.6 | f | f | 10.4 | f | 450 | 7.53 |
| Phase 3 | f | f | 13.5 | f | f | 12.2 | f | f | 11.1 | f | f | 10.5 | 580 | 7.88 |
| Short fallow winter cereal (wheat–barley) | *0.4* | f | 13.8 | f | 9.9 | f | 11.7 | f | 8.8 | f | 9.9 | f | 750 | 9.08 |
| Response cropping | | | | | | | | | | | | | | |
| Winter cereal–mung bean | *0.6* | 3.1 | 5.3 | 2.5 | 7.3 | 1.8 | f | 2.2 | 9.5 | 3.1 | 7.8 | 2.9 | 1050 | 7.68 |
| Sorghum–winter pulse | f | 11.3 | 2.5 | 9.4 | *0.4* | 10.1 | f | 7.2 | *1.4* | 11.0 | f | 11.7 | 1040 | 10.82 |
| Perennial pasture | | | | | | | | | | | | | | |
| Lucerne | f | f | 2.5 | 6.1 | 1.6 | 5.5 | 0.8 | 1.0 | 4.5 | 1.7 | 5.2 | 2.3 | 1700 | 5.20 |
| Lucerne + phalaris | f | f | 3.8 | 4.4 | 1.9 | 5.6 | 0.9 | 1.0 | 4.5 | 2.2 | f | f | 1340 | 4.86 |
| C3 + C4 grasses | f | Present[A] | 4.6 | 8.6 | 1.4 | 5.2 | 1.2 | 2.5 | 4.4 | 1.7 | 0.4 | 3.6 | 1820 | 5.60 |

[A]Young plants present but were not harvested.

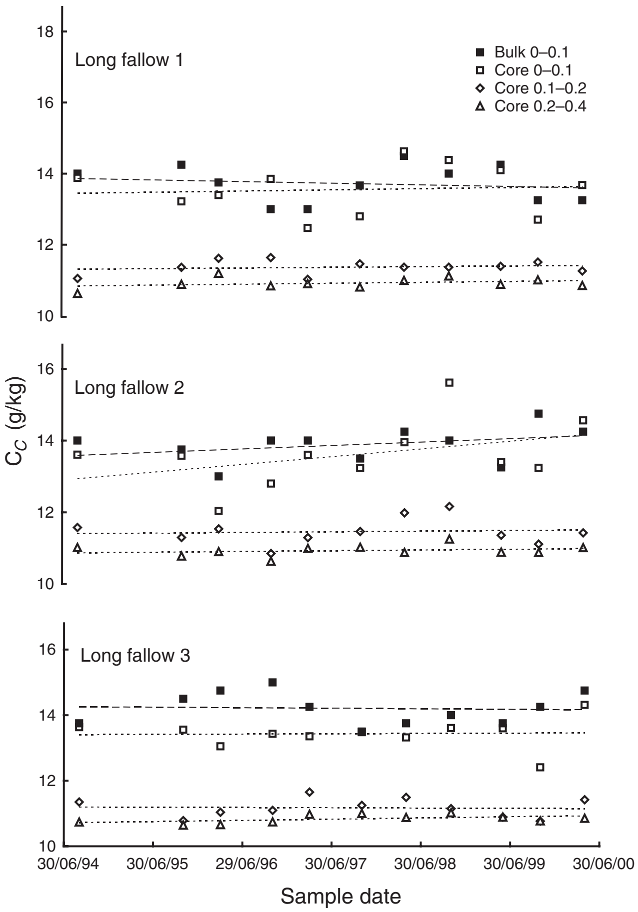


**Fig. 1.** No significant change in concentration of soil carbon ($C_C$) under zero tillage long fallow rotations on the Hudson field experiment. Bulked 0–0.1 m samples are shown as solid squares; individual core samples from 0–0.1, 0.1–0.2, 0.2–0.4 m depth increments are shown as open squares, diamonds, and triangles, respectively; solid lines show significant changes in $C_C$ over time, dashed lines are not significant. Rates of change of $C_S$ are shown in Table 2.

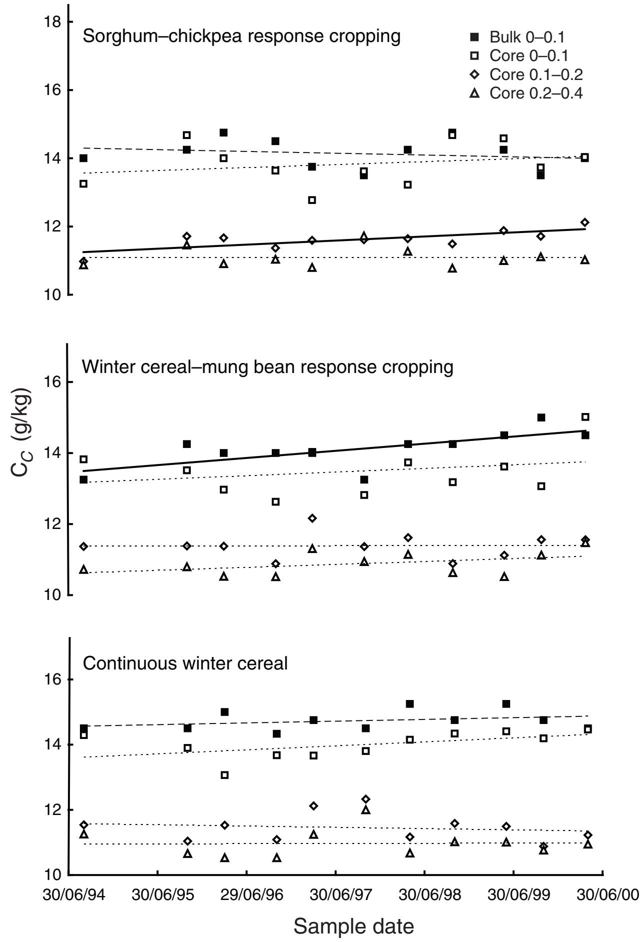


**Fig. 2.** No significant change in concentration of soil carbon ($C_C$) under zero tillage continuous winter cereal and some inconsistent evidence of accumulation under zero tillage response cropping on the Hudson field experiment. Bulked 0–0.1 m samples are shown as solid squares; individual core samples from 0–0.1, 0.1–0.2, 0.2–0.4 m depth increments are shown as open squares, diamonds, and triangles, respectively; solid lines show significant changes in $C_C$ over time, dashed lines are not significant. Rates of change of $C_S$ are shown in Table 2.

Over all crop and pasture systems on Hudson, there was an inverse relationship between annual aerial biomass yield and C accumulation rate (Fig. 4), suggesting that other processes (additions to soil C by roots and rhizodeposition, losses of C via root and microbial respiration) were occurring at different rates under different systems. Plant residence time had a stronger positive association with $C_S$. The long residence time of sorghum–chickpea response cropping was due to the sorghum harvest being delayed until the crop was frosted, sometimes 1–2 months after grain fill. During this time the sorghum was alive but producing little biomass.

### *Resampling of farm paddocks*

Bulk density ($\rho_b$) at the prevailing field soil water content ($\theta_g$) of soils from the paddocks sampled in May 2007 increased with depth from $\leq$1 near the surface to ~1.25 at 0.4–0.6 m depth (Fig. 5) and generally decreased with increasing $\theta_g$. [Average field $\theta_g$ (0–0.2 m) was dry (33–35 g/g) and similar for all land uses at Yarramanbah Creek. At Big Jacks Creek, cropped areas tended to be moist (40 g/g) while soils under native vegetation were drier (30 g/g).] Depth, site, land use, and most interactions together with $\theta_g$ at sampling significantly affected $\rho_b$. However, the significantly lower $\rho_b$ of surface layers under native vegetation compared with cropping soils was not responsive to moisture content within the range of $\theta_g$ encountered and so was treated as a land use effect independent of soil moisture content and was built into the reference profile for native vegetation.

The $C_C$ was largest in the surface soil under native vegetation and least at depth under Yarramanbah Creek cropping (Fig. 5). Overall, $C_C$ was less at Yarramanbah Creek than Big Jacks Creek. There was a well developed carbon profile under native vegetation with $C_C$ 6 times larger at the surface than at 0.4–0.6 m, whereas cropping and

**Table 2. Rates of change of soil carbon stocks under zero tillage cropping systems and perennial pastures on the Hudson field experiment**
Predicted value (PV) is the rate of change of soil carbon stock ($C_S$, Mg/ha.year). The standard error (s.e.) of each slope shows that slope's difference from zero. The overall standard error (s.e.d.) shows difference between slopes. PVs that are significantly different from zero are **bold**. No slopes that were significantly from zero were significantly different from each other

| System | 0–0.1 m bulked ($n$ = 394) | | 0–0.1 m core ($n$ = 863) | | 0.1–0.2 m core ($n$ = 860) | | 0.2–0.4 m core ($n$ = 861) | |
|---|---|---|---|---|---|---|---|---|
| | PV | s.e. | PV | s.e. | PV | s.e. | PV | s.e. |
| Long fallow wheat–sorghum rotations | | | | | | | | |
| Phase 1 | –0.094 | 0.078 | 0.024 | 0.141 | 0.022 | 0.066 | 0.029 | 0.055 |
| Phase 2 | 0.092 | 0.078 | 0.224 | 0.141 | 0.020 | 0.066 | 0.023 | 0.055 |
| Phase 3 | 0.050 | 0.077 | –0.003 | 0.141 | –0.010 | 0.066 | 0.042 | 0.055 |
| Short fallow winter cereal | 0.048 | 0.078 | 0.122 | 0.141 | –0.036 | 0.066 | 0.008 | 0.055 |
| Response cropping | | | | | | | | |
| Winter cereal–mung bean | **0.208** | 0.082 | 0.099 | 0.141 | 0.003 | 0.066 | 0.092 | 0.055 |
| Sorghum–winter pulse | –0.054 | 0.078 | 0.082 | 0.141 | **0.134** | 0.066 | –0.001 | 0.055 |
| Perennial pasture | | | | | | | | |
| Lucerne | **0.191** | 0.078 | **0.379** | 0.141 | **0.286** | 0.066 | **0.122** | 0.055 |
| Lucerne + phalaris | **0.350** | 0.079 | **0.407** | 0.141 | **0.165** | 0.066 | 0.046 | 0.055 |
| C3 + C4 grasses | **0.313** | 0.078 | **0.484** | 0.141 | **0.147** | 0.066 | 0.054 | 0.055 |
| s.e.d. of system slopes | | 0.108 | | 0.188 | | 0.089 | | 0.073 |

pasture returned to cropping at both sites showed depleted profiles, with $C_C$ at the surface barely twice that at depth.

### *Comparison of farm paddocks over time*

There were no significant changes in $C_S$ over time under zero tillage cropping at either site on the commercial properties (Table 3), corroborating the results from the cropping systems on the Hudson experiment. Date of sampling and interactions of date with other fixed effects were not significant. Neither was there evidence of accumulation of $C_S$ under zero tillage cropping, or evidence of $C_S$ loss after zero tillage cropping of old pastures. $C_S$ at 0.4–0.6 m depth remained unchanged under all land uses at both sites, clearly justifying the use of 0.4–0.6 m data from the Hudson experiment to correct for laboratory drift.

## Discussion

These data suggest that dryland farmers on north-western NSW Vertosols, where annual rainfall <700 mm, cannot expect to accumulate soil carbon in continuously cropped land, even under zero tillage, at least within the short to medium term. The lack of accumulation of $C_S$ under cropping, either experimentally on Hudson or on a limited number of Liverpool Plains farms (including the highly productive Big Jacks Creek continuous cropping paddocks, which were 30% more productive than all the other paddocks and the Hudson experiment), agrees with reports from SE Queensland (Dalal *et al*. 1995), the Central Highlands of Queensland (Armstrong *et al*. 2003), and more generally in eastern Australia (Dalal and Chan 2001; Chan *et al*. 2003). All found that carbon sequestration under continuous zero tillage cropping was negligible.

The significant accumulation of $C_S$ under perennial pastures on Hudson and elsewhere (Dalal *et al*. 1995; Dalal and Chan 2001) indicates that soil carbon in cropping systems might be increased by inclusion of periods of perennial pasture in rotation with annual crops. Unchanged $C_S$ after conversion of pasture to zero tillage cropping on the nearby farm sites indicates that zero tillage cropping in this environment may at least conserve the $C_S$ accumulated under pasture. Therefore, a net accumulation of $C_S$ might occur under 5–6-year sequences of healthy perennial pasture in rotation with zero tillage cropping.

Despite our results and those from Queensland, there is nevertheless evidence in the literature for increased $C_S$ under zero tillage systems in Australia. For example, on a fertile red earth in south-eastern NSW, $C_S$ increased under a zero tillage wheat–subterreanean clover rotation and remained unchanged (~20 Mg/ha, 0–0.1 m) over 20 years under zero tillage wheat–lupin (Heenan *et al*. 2004). This disparity might be explained by the relatively large wheat biomass yields (14.5 Mg/ha) realised in the south, together with other factors, such as the drier southern summers leading to lower rates of microbial respiration compared with summer rainfall environments.

We might expect that net accumulation of $C_S$ under zero tillage seasonal cropping would occur most rapidly in extremely carbon-depleted soils (Stewart *et al*. 2007) as suggested for parts of North America (Follett 2001). For example, if high rates of biomass production are achieved relative to losses of labile carbon due to respiration and erosion, it might be possible to increase total soil carbon. Carbon accumulation may also be possible with relatively modest rates of biomass production if a significant proportion of photosynthate is channelled below ground. However, there is scant evidence for such processes occurring in annual cropping systems in semi-arid eastern Australia. Even where Vertosols in Queensland were described as depleted ($C_C$ 6–7 g/kg) and had been under conventional cropping for many decades (Dalal and Mayer 1986; Dalal *et al*. 1995; Armstrong *et al*. 2003), accumulation of soil C was found only under perennial pastures, with none being observed under zero tillage cropping. Our apparently inverse

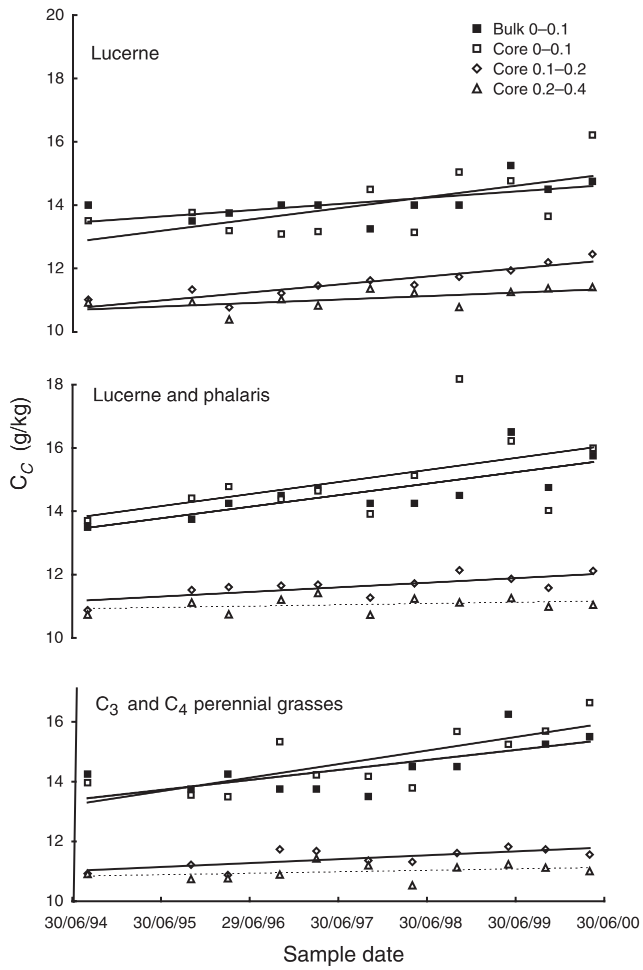


**Fig. 3.** Good evidence of significant accumulation of soil carbon concentration ($C_C$) under perennial pastures down to 0.2 m depth on the Hudson field experiment. Bulked 0–0.1 m samples are shown as solid squares; individual core samples from 0–0.1, 0.1–0.2, 0.2–0.4 m depth increments are shown as open squares, diamonds, and triangles, respectively; solid lines show significant changes in $C_C$ over time, dashed lines are not significant. Rates of change of $C_S$ are shown in Table 2.

relationship between aerial biomass production and rates of $C_S$ accumulation (Fig. 4*a*) suggests caution when making general comparisons of quite different systems. The effects of aerial biomass production appear to be less important than other factors, which were reasonably represented by residence time (Fig. 4*b*). This is illustrated by a simple carbon balance (Table 4) derived from Hudson data and reported shoot/root values (1/0.6; Johnson *et al*. 2006; Teixeira *et al*. 2008). In this analysis, cropping is characterised by large aerial, and hence root and rhizosphere, biomass C values. However, fallow periods, most likely, contributed to net losses of most (response cropping) or all (long and short fallow cropping) of labile residue and soil carbon. Although pastures yielded slightly less total C than crops, total residues were only about half those of crops, due to export of forage. The significant measured accumulation of C under pastures compared with cropping is shown in Table 4 as having been due to lower rates of C loss under pasture (below ground C was calculated in the same way for both crops and pastures as 60% of measured aerial C). Respiration losses may have been lower under perennial pastures due to their generally rapid response to rainfall in this environment, so keeping the soil profile much drier and for longer periods compared with soils under short and long fallow cropping sequences (Ringrose-Voase *et al*. 2003). In addition, larger rates of C accumulation as root and rhizosphere deposition (together with concomitantly larger rates of loss than those calculated in Table 4) in the surface layers of the soil may have occurred under pastures compared with cropping due to the restriction of most pasture root growth and rhizosphere activity to the intermittently moist surface layers. The high plant-available water-holding capacity of these Vertosols (100 mm in the 0–0.4 m depth layers) retains most rainfall near the soil surface when the soil is being continually dried by perennial vegetation. However, it is likely that root biomass forms a greater proportion of total perennial pasture biomass. For example, in southern Queensland (Dalal *et al*. 1995) pasture root mass, measured down to 1.5 m, was found to be double that of its aerial biomass, whereas wheat root biomass was only ~40% of total aerial biomass. Furthermore, the ability of pastures to sequester soil carbon may be due not only to the probably greater proportion of total biomass carbon as root carbon in perennial pastures compared with annual crops but also to the longer residence time in soil of that root carbon compared with shoot carbon (Rasse *et al*. 2005). The significant rates of accumulation of $C_S$ under perennial pastures, compared with annual crops with larger yields of aerial biomass, are most probably due to a combination of lower rates of loss of soil C, especially during dry periods, and deposition of larger proportions of photosynthate below ground during periods of growth.

The Vertosols studied here have distinctive characteristics, such as high clay content (75–80%) with a large proportion of smectite (90%) conferring considerable shrink–swell reactivity to changes in water content, which contributes to their generally high water-holding capacity and ability to recover from compaction. With appropriate management, cropped Vertosols with depleted nutrient and organic matter can be particularly productive in terms of biomass (Young *et al*. 2008) and therefore have the potential to accumulate soil carbon. Although studies in north-western NSW (Young *et al*. 2005; Wilson *et al*. 2008) on a range of soil types have demonstrated that the quantity of carbon stored in Vertosols typically exceeds that of lighter textured soils, the relative differences between land-uses follows the same trend across most soil types.

Internationally, there is considerable evidence for carbon accumulation under zero tillage. However, such results typically have been observed in climates that are moister and sometimes cooler than those of the rainfed cereal cropping areas of Australia. For example, in the warm, moist (average annual rainfall >1000 mm) environment of south-eastern USA, conservation tillage is seen as an effective strategy to regain $C_S$ lost after decades, and sometimes centuries, of intensive tillage and erosion. Using traditional tillage as a baseline, tillage research in that region (Franzluebbers 2005) has

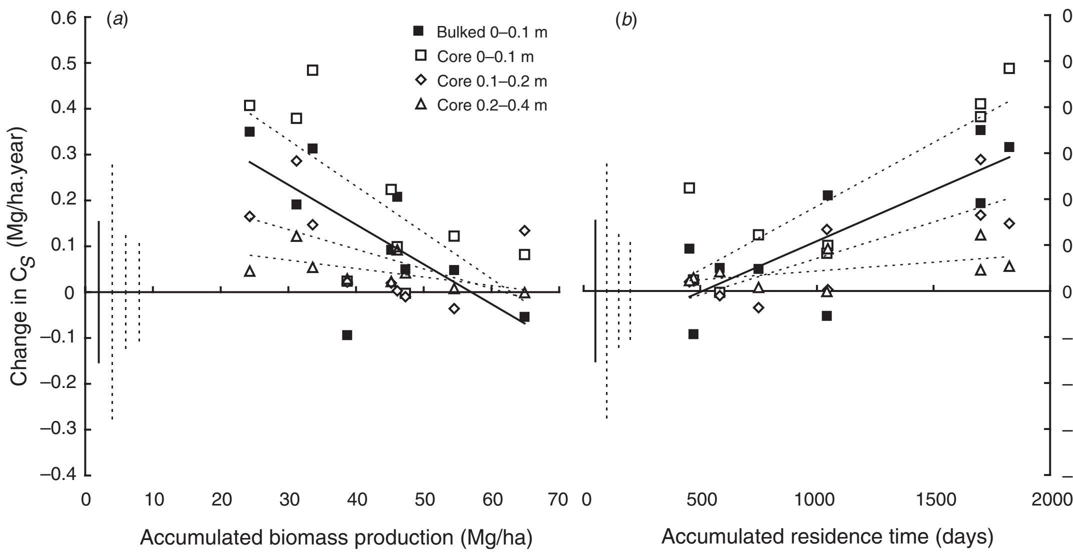


**Fig. 4.** Rates of change of soil carbon stocks ($C_S$) in annual cropping and perennial pasture systems on the Hudson field experiment (Table 2) appear (*a*) inversely proportional to accumulated aerial biomass production and (*b*) directly proportional to the accumulated residence times of established live plants. A rate of $C_S$ accumulation value needs to be above or below the relevant bar to be different from zero; the bars, from left to right on each graph, refer to the bulked 0–0.1 m samples (shown as solid squares with solid fitted line and solid bar) and the individual core samples from 0–0.1, 0.1–0.2, 0.2–0.4 m depth increments (shown as open squares, diamonds, and triangles with dashed fitted lines and dashed bars, respectively). The respective correlation coefficients from speculative linear regressions are: (*a*) 0.49, 0.49, 0.25, 0.34; and (*b*) 0.63, 0.73, 0.67, 0.28.

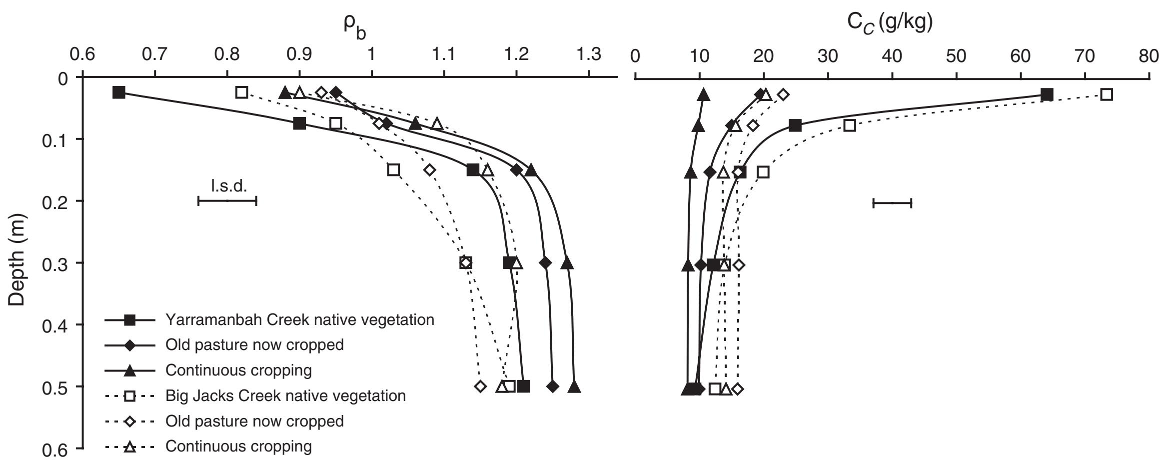


**Fig. 5.** Bulk density ($\rho_b$, Mg/m$^3$) at field soil water content ($\theta_g$) near the soil surface was significantly less (least significant difference, l.s.d., shown by horizontal bar) in grassy woodland (square symbols) compared to cropping soils (diamonds and triangles). (The l.s.d. applies only to $\rho_b$ at the prevailing values of $\theta_g$ at the time of sampling; the values of $\rho_b$ shown here occur within a range of ~0.1 Mg/m$^3$, depending on $\theta_g$.) Soil carbon concentration ($C_C$) was significantly larger near the soil surface in grassy woodland than in cropping soils. Most means were not significantly different in the 0.4–0.6 m depth increment. This dataset was derived from 86 cores.

shown an average rate of accumulation of $C_S$ with zero tillage of 0.42 Mg/ha.year over an average observation period of 10 years. By increasing cropping complexity (intensity and diversity) and adding N fertiliser, $C_S$ could be increased by ~0.25 Mg/ha.year, irrespective of tillage management. $C_S$ accumulation under old cropping land converted to forages was greater and averaged 1.0 Mg/ha over an average of 15 years. A similar picture is emerging from work in central Brazil (average annual rainfall >1000 mm) where the mean rate of $C_S$ accumulation under no-tillage summer cropping in tropical soils was estimated to be 0.35 Mg/ha.year over 4–20 years, whereas in southern Brazilian subtropical soils where both summer and winter cropping are practised, estimated mean $C_S$ accumulation was 0.48 Mg/ha.year over 9–22 years (Bayer *et al*. 2006). In the Brazilian environment, accumulation of surface $C_S$ under no-tillage may actually

**Table 3. No short-term changes in carbon stocks ($C_S$, t/ha) in Liverpool Plains Vertosols under zero tillage cropping after traditional cropping or perennial pasture**

All site.landuse cells were a single farmer's paddock or block except Big Jacks Creek continuous cropping, which consisted of 4 blocks with similar histories; the dataset was derived from 175 cores with 8–21 cores/site.landuse cell. Aggregated Hudson data are shown for comparison. Change from traditional to zero tillage cropping occurred at Yarramanbah Creek around 2001 and at Big Jacks Creek from 1990 to 1994. Yarramanbah Creek pasture paddock was successfully sown to lucerne in late 1980s after 7 years of cropping; Big Jacks Creek pasture paddock successfully sown to lucerne and phalaris in 1991 after 3 years of cropping; native vegetation was 1–2-ha remnants of grassy eucalypt woodland. Between the times of sampling, winter cereal and grain sorghum were grown at both sites, faba bean was grown in the continuous cropping sequence at Big Jacks Creek in 2000 and at Yarramanbah Creek in the 'old pasture' in 2006. Biomass (oven-dry weight) was estimated from grain yield (assuming 12% moisture content) assuming a harvest index of 45% for crops yielding <5 t/ha, 48% for crops yielding >5 t/ha (Young *et al.* 2008). Carbon stock ($C_S$) was calculated from concentration and reference profile BD (0–0.2 m, 1.003 (cropping) and 0.977 (native vegetation); 0.2–0.4 m, 1.121; 0.4–0.6 m, 1.144). Date of sampling and date interactions with site, land use, and depth were not significant. Site, land use, depth, and their interactions were significant

| Site | Land use | | Depth (m) | 1998–2000 $C_S$ | 1998–2000 s.e. | May 2007 $C_S$ | May 2007 s.e. |
|---|---|---|---|---|---|---|---|
| Yarramanbah Creek | Native vegetation | Intermittent grazing | 0–0.2 | 38.3 | 2.7 | 50.2 | 2.1 |
| | | | 0.2–0.4 | 24.4 | 1.1 | 27.0 | 0.8 |
| | | | 0.4–0.6 | 20.5 | 1.2 | 21.2 | 0.9 |
| | Old pasture now cropped (crops/year) | 1 | 0–0.2 | 27.2 | 2.4 | 28.5 | 2.3 |
| | N application (kg/ha.year) | 100 | 0.2–0.4 | 23.7 | 1.0 | 22.9 | 0.9 |
| | Biomass (Mg/ha.year) | 8 | 0.4–0.6 | 21.9 | 1.1 | 22.5 | 1.0 |
| | Continuous cropping (crops/year) | 1 | 0–0.2 | 18.3 | 2.2 | 20.1 | 2.0 |
| | N application (kg/ha.year) | 100 | 0.2–0.4 | 18.9 | 0.9 | 18.4 | 0.8 |
| | Biomass (Mg/ha.year) | 8 | 0.4–0.6 | 18.9 | 1.0 | 18.5 | 0.9 |
| Big Jacks Creek | Native vegetation | Intermittent grazing | 0–0.2 | 57.5 | 3.1 | 64.3 | 2.7 |
| | | | 0.2–0.4 | 32.1 | 1.3 | 31.2 | 1.1 |
| | | | 0.4–0.6 | 29.9 | 1.4 | 28.4 | 1.2 |
| | Old pasture now cropped (crops/year) | 1 | 0–0.2 | 38.6 | 3.3 | 36.0 | 3.3 |
| | N application (kg/ha.year) | 105 | 0.2–0.4 | 34.7 | 1.3 | 36.1 | 1.3 |
| | Biomass (Mg/ha.year) | 8 | 0.4–0.6 | 33.9 | 1.5 | 36.2 | 1.5 |
| | Continuous cropping (crops/year) | 1.1 | 0–0.2 | 31.7 | 2.2 | 30.2 | 2.2 |
| | N application (kg/ha.year) | 115 | 0.2–0.4 | 31.4 | 0.9 | 30.9 | 0.9 |
| | Biomass (Mg/ha.year) | 11 | 0.4–0.6 | 31.4 | 1.0 | 32.2 | 1.0 |
| | | | | l.s.d. = 4.76 | | | |
| Hudson | | | | 1994 | | 2000 | |
| | Continuous cropping (crops/year) | 1 | 0–0.2 | 26.5 | | 27.3 | |
| | N application (kg/ha.year) | 90 | 0.2–0.4 | 24.2 | | 24.5 | |
| | Biomass (Mg/ha.year) | 8 | | | | | |
| | Perennial pasture biomass (Mg/ha.year) | 5 | 0–0.2 | 26.1 | | 29.9 | |
| | N application to C3 + C4 grasses (kg/ha.year) | 50 | 0.2–0.4 | 24.1 | | 24.8 | |

**Table 4. Average annual carbon balance of zero tillage cropping systems and perennial pastures on the Hudson field experiment**

All data are rates of carbon flux expressed as t/ha.year. Measured data are shown in regular font, derived data are *italicised*. Export is harvested and removed grain C from crops and forage C from pastures. Aerial residue of crops is final harvest total biomass C less grain C; for pastures it is detached plant material on the soil surface (litter). Pasture roots include crowns from which shoots have been harvested (Teixeira *et al.* 2008). For calculation of root and rhizosphere deposition, C in 0–0.3 m surface soil: shoot/root 1/0.6 (Johnson *et al.* 2006; a slightly larger value than that reported for lucerne taproots (~1/0.5) by Teixeira *et al.* 2008); organic matter 40% C (Johnson *et al.* 2006)

| System | *Total yield (T)* | Export (E) | Aerial residue (AR) | *Roots + rhizosphere deposition (RRD)* | *Total residue (TR)* | *Loss (L)* | Net accumulation 0–0.3 m soil (NA) |
|---|---|---|---|---|---|---|---|
| Calculation: | *E + AR + RRD* | Measured | | *0.6[E + AR]* | *AR + RRD* | *TR – NA* | Measured |
| Long fallow 1 | *4.46* | 1.27 | 1.52 | *1.67* | *3.19* | *3.19* | 0.00 |
| Long fallow 2 | *5.26* | 1.29 | 2.00 | *1.97* | *3.97* | *3.97* | 0.00 |
| Long fallow 3 | *5.50* | 1.23 | 2.20 | *2.06* | *4.27* | *4.27* | 0.00 |
| Short fallow winter cereal | *6.29* | 1.45 | 2.48 | *2.36* | *4.83* | *4.83* | 0.00 |
| Winter cereal–mung bean | *5.29* | 1.09 | 2.22 | *1.98* | *4.20* | *4.10* | **0.10** |
| Sorghum–winter pulse | *7.52* | 1.64 | 3.06 | *2.82* | *5.89* | *5.76* | **0.13** |
| Lucerne | *4.25* | 2.50 | 0.16 | *1.59* | *1.75* | *1.06* | **0.69** |
| Lucerne + phalaris | *4.25* | 2.43 | 0.23 | *1.59* | *1.82* | *1.27* | **0.54** |
| C3 + C4 grasses | *4.67* | 2.69 | 0.23 | *1.75* | *1.98* | *1.43* | **0.55** |

exceed that of the native Cerrado soils due to application of lime, mitigation of nutrient deficiencies, the addition of N, and the inclusion of winter legume cover crops (Diekow *et al*. 2005; Zanatta *et al*. 2007). However, without legume cover crops or regular N additions in tropical central Brazil, rates of $C_S$ depletion under zero tillage were 10 Mg/ha (0–1 m) over 20 years of a largely maize/soybean–winter fallow sequence but were 3-fold less than that under tillage (Jantalia *et al*. 2007).

The accumulation of $C_S$ in these warm, moist climates could be attributed to greater biomass addition and more frequent double cropping than is achievable in inland eastern Australian under rainfed conditions. Unfortunately, few data on crop production are reported where the focus was on soil carbon. In one exception (Bayer *et al*. 2006), significant accumulation of $C_S$ was reported under no-tillage summer cropping on previously intensely cultivated tropical soils in central Brazil where average annual residue addition was ~4 Mg/ha. This is less than we measured under continuous winter cereal or sorghum–chickpea response cropping (both ~5 Mg/ha.year) on Hudson.

In the 'corn belt' of moist temperate central USA (average annual rainfall 500–1300 mm, $C_C$ under original prairie grassland, 40 g/kg), the rate of storage of $C_S$ under no-tillage compared with traditional tillage has been significant but variable, averaging 0.4 Mg/ha.year over 15–30 years (Johnson *et al*. 2005). Where cropping land has been returned to perennial grass, the accumulation rate has been larger, 0.56 Mg/ha.year. The attributes of this environment, compared with the semi-arid cropping regions of eastern Australia, that permit a positive balance between primary production and net decomposition of carbon may be the synchrony of climatic conditions. Similar seasonal distribution of temperature and rainfall produce corn biomass of up to 20 Mg/ha, leaving residues of ~10 Mg/ha (considerably larger than the 5.4 Mg/ha required to maintain $C_S$ in that environment; Wilhelm *et al*. 2004), and cold winters reducing microbial respiration (Huggins *et al*. 1998). However, although losses of soil carbon due to respiration during winter fallow were generally low in colder, semi-arid Canadian (Franzluebbers and Arshad 1996; Deen and Kataki 2003) and moist Swiss (Hermle *et al*. 2008) environments, the rate of incorporation of crop residues into soil carbon was also low, producing an overall net loss of soil carbon under zero tillage.

It appears that under dryland cropping in much of eastern Australia, with the possible exception of some higher rainfall slopes regions (Heenan *et al*. 2004), either seasonal rainfall is insufficient to produce the required biomass, or microbial respiration too often exceeds carbon fixation due to warm and moist fallow periods, for net accumulation of $C_S$ under annual cropping. However, plant attributes may play a significant part in enhancing $C_S$ (De Deyn *et al*. 2008) and could be significant when conditions are marginal for net accumulation of $C_S$. For example, there is significant accumulation of $C_S$ under Hudson pastures and in SE Queensland (Dalal *et al*. 1995) despite the removal of aerial biomass. An explanation for the accumulation of $C_S$ under the Hudson sorghum–chickpea (0.1–0.2 m) might be the enhanced accumulation of $C_S$ from the symbiotic arbuscular mycorrhiza (Langley and Hungate 2003) associated with sorghum roots (Thompson 1987). Although aerial biomass of grain and pasture legumes was less than that of N fertilised grass species, $C_S$ under legumes may be enhanced by the channelling of photosynthate to the atmospheric N fixing symbiosis.

## Conclusion

Current knowledge strongly suggests that dryland farmers in north-western NSW cannot expect to accumulate soil carbon in continuously cropped land within the short–medium term. The inclusion of healthy perennial pastures in rotation with crops may assist in a slow net accumulation of carbon, although this has not been demonstrated over the medium to long-term. Some crop and pasture plants might have growth and symbiotic characteristics that could be used to enhance $C_S$ but this requires further work. For carbon trading purposes, woodland systems are likely to sequester more carbon than improved management of cropping systems. Although it has not been demonstrated that soil carbon accumulates under zero tillage cropping on Vertosols on the Liverpool Plains, the increased financial returns and soil and water conservation benefits from these much improved practices are now widely recognised.

## Acknowledgments

We thank Robert and Edwina Duddy for making the site on 'Hudson' available for our work, and Neil Barwick, Brien Cobcroft, and James Badgery for access to their country. We thank Alison Bowman, Brendan George, Yin Chan, and 2 anonymous referees for very helpful comments on earlier drafts, Anthony Ringrose-Voase for his insightful comments on the bulk density calculations, and Ross McLeod and Wayne McPherson for their excellent work in the paddock and in the soil processing shed. This work was funded by a NSW Government Climate Action Grant (T06/CAG/003) made avalilable to NSW Department of Primary Industries on the recommendation of the Namoi Catchment Management Authority. The field experiment and original sampling of farmers' paddocks was funded by a series of grants from GRDC, Salt Action and Land and Water Australia from 1993 to 2002.

## References

Armstrong RD, Millar G, Halpin NV, Reid DJ, Standley J (2003) Using zero tillage, fertilisers and legume rotations to maintain productivity and soil fertility in opportunity cropping systems on a shallow Vertosol. *Australian Journal of Experimental Agriculture* **43**, 141–153. doi: 10.1071/EA01175

Banks R (1998) 'Soil landscapes of the Blackville 1 : 100 000 Sheet Report.' (Department of Conservation and Land Management: Sydney)

Bayer C, Martin Neto L, Mielniczuk J, Pavinato A, Dieckow J (2006) Carbon sequestration in two Brazilian Cerrado soils under no-till. *Soil & Tillage Research* **86**, 237–245. doi: 10.1016/j.still.2005.02.023

Butler DG, Cullis BR, Gilmour AR, Gogel BJ (2007) 'ASReml-R Reference Manual Release 2.' (Queensland Department of Primary Industries)

Chan KY, Heenan DP, So HB (2003) Sequestration of carbon and changes in soil quality under conservation tillage on light-textured soils in Australia: a review. *Australian Journal of Experimental Agriculture* **43**, 325–334. doi: 10.1071/EA02077

Dalal RC (1989) Long-term effects of no-tillage, crop residue, and nitrogen application on properties of a Vertisol. *Soil Science Society of America Journal* **53**, 1511–1515.

Dalal RC, Chan KY (2001) Soil organic matter in rainfed cropping systems of the Australian cereal belt. *Australian Journal of Soil Research* **39**, 435–464. doi: 10.1071/SR99042

Dalal RC, Mayer RJ (1986) Long-term trends in fertility of soils under continuous cultivation and cereal cropping in southern Queensland. II. Total organic carbon and its rate of loss from the soil profile. *Australian Journal of Soil Research* **24**, 281–292. doi: 10.1071/SR9860281

Dalal RC, Strong WM, Weston EJ, Cooper JE, Lehane KJ, King AJ, Chicken CJ (1995) Sustaining productivity of a Vertisol at Warra, Queensland, with fertilisers, no-tillage, or legumes. 1. Organic matter status. *Australian Journal of Experimental Agriculture* **35**, 903–913. doi: 10.1071/EA9950903

De Deyn GB, Cornelissen JHC, Bardgett RD (2008) Plant functional traits and soil carbon sequestration in contrasting biomes. *Ecology Letters* **11**, 516–531. doi: 10.1111/j.1461-0248.2008.01164.x

Deen W, Kataki PK (2003) Carbon sequestration in a long-term conventional versus conservation tillage experiment. *Soil & Tillage Research* **74**, 143–150. doi: 10.1016/S0167-1987(03)00162-4

Department of Climate Change (2008) National Greenhouse Gas Inventory 2006. Department of Climate Change, Commonwealth of Australia, Canberra.

Diekow J, Mielniczuk J, Knicker H, Bayer C, Dick DP, Kogel-Knabner I (2005) Carbon and nitrogen stocks in physical fractions of a subtropical Acrisol as influenced by long-term no-till cropping systems and N fertilisation. *Plant and Soil* **268**, 319–328. doi: 10.1007/s11104-004-0330-4

Ellert BH, Bettany JR (1995) Calculation of organic matter and nutrients stored in soils under contrasting management regimes. *Canadian Journal of Soil Science* **75**, 529–538.

Fettell NA, Gill HS (1995) Long-term effects of tillage, stubble, and nitrogen management on properties of a red-brown earth. *Australian Journal of Experimental Agriculture* **35**, 923–928. doi: 10.1071/EA9950923

Follett RF (2001) Soil management concepts and carbon sequestration in cropland soils. *Soil & Tillage Research* **61**, 77–92. doi: 10.1016/S0167-1987(01)00180-5

Franzluebbers AJ (2005) Soil organic carbon sequestration and agricultural greenhouse gas emissions in the southeastern USA. (Special Issue: Greenhouse gas contributions and mitigation potential in agricultural regions of North America.) *Soil & Tillage Research* **83**, 120–147.

Franzluebbers AJ, Arshad MA (1996) Soil organic matter pools with conventional and zero tillage in a cold, semiarid climate. *Soil & Tillage Research* **39**, 1–11. doi: 10.1016/S0167-1987(96)01055-0

Gardner EA (1988) Soil water. In 'Understanding soils and soil data'. (Ed. IF Fergus) (Australian Society of Soil Science (Queensland Branch): Brisbane, Qld)

Gilmour AR, Cullis B, Gogel BJ, Welham S, Thompson R (2006) 'ASReml User Guide. Release 2.' (VSN International: UK)

Heenan DP, Chan KY, Knight PG (2004) Long-term impact of rotation, tillage and stubble management on the loss of soil organic carbon and nitrogen from a Chromic Luvisol. *Soil & Tillage Research* **76**, 59–68. doi: 10.1016/j.still.2003.08.005

Heenan DP, McGhie WJ, Thomson FM, Chan KY (1995) Decline in soil organic carbon and total nitrogen in relation to tillage, stubble management, and rotation. *Australian Journal of Experimental Agriculture* **35**, 877–884. doi: 10.1071/EA9950877

Hermle S, Anken T, Leifeld J, Weisskopf P (2008) The effect of the tillage system on soil organic carbon content under moist, cold-temperate conditions. *Soil & Tillage Research* **98**, 94–105. doi: 10.1016/j.still.2007.10.010

Holford IC (1981) Changes in nitrogen and organic carbon of wheat-growing soils after various periods of grazed lucerne, extended fallowing and continuous wheat. *Australian Journal of Soil Research* **19**, 239–249. doi: 10.1071/SR9810239

Huggins DR, Buyanovsky GA, Wagner GH, Brown JR, Darmody RG, Peck TR, Lesoing GW, Vanotti MB, Bundy LG (1998) Soil organic C in the tallgrass prairie-derived region of the corn belt: effects of long-term crop management. *Soil & Tillage Research* **47**, 219–234. doi: 10.1016/S0167-1987(98)00108-1

IPCC (2007) Climatic Change 2007. Synthesis Report. Intergovernmental Panel on Climate Change Fourth Assessment Report, IPCC, Geneva.

Isbell RF (1996) 'The Australian Soil Classification.' (CSIRO Publishing: Collingwood, Vic.)

Jantalia CP, Resck DV, Alves BJ, Zotarelli L, Urquiaga S, Boddey RM (2007) Tillage effect on C stocks of a clayey Oxisol under a soybean-based crop rotation in the Brazilian Cerrado region. *Soil & Tillage Research* **95**, 97–109. doi: 10.1016/j.still.2006.11.005

Johnson JM, Allmaras RR, Reicosky DC (2006) Estimating source carbon from crop residues, roots and rhizodeposits using the national grain-yield database. *Agronomy Journal* **98**, 622–636. doi: 10.2134/agronj2005.0179

Johnson JM, Reicosky DC, Allmaras RR, Sauer TJ, Venterea RT, Dell CJ (2005) Greenhouse gas contributions and mitigation potential of agriculture in the central USA. (Special Issue: Greenhouse gas contributions and mitigation potential in agricultural regions of North America.) *Soil & Tillage Research* **83**, 73–94.

Kirby JM, Bernardi AL, Ringrose-Voase AJ, Young R, Rose H (2003) Field swelling, shrinking, and water content change in a heavy clay soil. *Australian Journal of Soil Research* **41**, 963–978. doi: 10.1071/SR02055

Lal R, Follett RF, Kimble JM (2003) Achieving soil carbon sequestration in the United States: a challenge to the policy makers. *Soil Science* **168**, 827–845. doi: 10.1097/01.ss.0000106407.84926.6b

Langley JA, Hungate BA (2003) Mycorrhoizal controls on belowground litter quality. *Ecology* **84**, 2302–2312. doi: 10.1890/02-0282

Paydar Z, Huth N, Ringrose-Voase A, Young R, Bernardi T, Keating B, Cresswell H (2005) Deep drainage and land use systems. Model verification and systems comparison. *Australian Journal of Agricultural Research* **56**, 995–1007. doi: 10.1071/AR04303

Rasse DP, Rumpel C, Dignac MF (2005) Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. *Plant and Soil* **269**, 341–356. doi: 10.1007/s11104-004-0907-y

Ringrose-Voase AJ, Young RR, Paydar Z, Huth NI, Bernardi AL, Cresswell HP, Keating BA, Scott JF, Stauffacher M, Banks RG, Holland JF, Johnston RM, Green TW, Gregory LJ, Daniells I, Farquharson R, Drinkwater RJ, Heidenreich S, Donaldson S (2003) Deep drainage under different land uses in the Liverpool Plains Catchment. Report 3, Agricultural Resource Management Report Series, NSW Agriculture, Tamworth, NSW.

Smith P, Powlson DS, Glendining MJ, Smith JU (1998) Preliminary estimates of the potential for carbon mitigation in European soils through no-till farming. *Global Change Biology* **4**, 679–685. doi: 10.1046/j.1365-2486.1998.00185.x

Stewart CE, Paustian K, Conant RT, Plante AF, Six J (2007) Soil carbon saturation: concept, evidence and evaluation. *Biogeochemistry* **86**, 19–31. doi: 10.1007/s10533-007-9140-0

Teixeira EI, Moot DJ, Brown HE (2008) Defoliation frequency and season affected radiation use efficiency and dry matter partitioning to roots of lucerne (*Medicago sativa* L.) crops. *European Journal of Agronomy* **28**, 103–111. doi: 10.1016/j.eja.2007.05.004

Thompson JP (1987) Decline of vesicular-arbuscular mycorrhizae in long fallow disorder of field crops and its expression in phosphorus deficiency of sunflower. *Australian Journal of Agricultural Research* **38**, 847–867. doi: 10.1071/AR9870847

Verbyla AP, Cullis BP, Kenward MG, Welham SJ (1999) The analysis of designed experiments and longitudinal data using smoothing splines (with discussion). *Applied Statistics* **48**, 269–311. doi: 10.1111/1467-9876.00154

Wang WJ, Dalal RC, Moody PW (2004) Soil carbon sequestration and density distribution in a Vertosol under different farming practices. *Australian Journal of Soil Research* **42**, 875–882. doi: 10.1071/SR04023

Wilhelm WW, Johnson JM, Hatfield JL, Voorhoees WB, Linden DR (2004) Crop and soil productivity response to corn residue removal: a literature review. *Agronomy Journal* **96**, 1–17.

Wilson BR, Growns I, Lemon J (2008) Land-use effects on soil carbon and other soil properties on the NW slopes of NSW: Implications for soil condition assessment. *Australian Journal of Soil Research* **46**, 359–367. doi: 10.1071/SR07231

Young R, Wilson BR, McLeod M, Alston C (2005) Carbon storage in the soils and vegetation of contrasting land uses in northern New South Wales, Australia. *Australian Journal of Soil Research* **43**, 21–31. doi: 10.1071/SR04032

Young RR, Derham P-J, Dunin FX, Bernardi AL, Harden S (2008) High crop productivity with high water use in winter and summer on the Liverpool Plains, eastern Australia. *Australian Journal of Agricultural Research* **59**, 303–320. doi: 10.1071/AR07138

Zanatta JA, Bayer C, Dieckow J, Vieira FC, Mielniczuk J (2007) Soil organic carbon accumulation and carbon costs related to tillage, cropping systems and nitrogen fertilization in a subtropical Acrisol. *Soil & Tillage Research* **94**, 510–519. doi: 10.1016/j.still.2006.10.003

Manuscript received 30 April 2008, accepted 2 December 2008