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Managing semi-arid woodlands for carbon storage: Grazing and shrub effects on above- and belowground carbon

Stefani Daryanto[a], David J. Eldridge[b,*], Heather L. Throop[c,1]

[a] Evolution and Ecology Research Centre, School of Biological, Earth and Environmental Sciences, University of New South Wales, Sydney, NSW 2052, Australia
[b] Office of Environment and Heritage, c/-Australian Wetlands, Rivers and Landscapes Centre, School of Biological, Earth and Environmental Sciences, University of New South Wales, Sydney, NSW 2052, Australia
[c] Biology Department, New Mexico State University, MSC 3AF, Las Cruces, NM 88003, USA

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ABSTRACT

Shrub cover has increased in semi-arid regions worldwide. This change has generally been viewed as land degradation, due to shrub-induced declines in pastoral productivity. As a consequence, widespread management treatments to reduce shrub density have been applied in many pastoral areas. These treatments, however, often do not have long-term positive benefits for forage production. Alternative uses for shrub-encroached lands have received little consideration, but a recent move towards economic incentives for carbon (C) storage could lead to financially viable alternative land management strategies. We examined changes in above- and belowground C storage following 20 years of factorial land management treatments (grazing/no grazing and shrub removal/no removal) in an Australian semi-arid woodland. Disturbance by shrub removal (root ploughing) and/or livestock grazing significantly reduced the amount of soil organic carbon (SOC). The most disturbed treatment (grazed and ploughed) contained the least SOC (15.30 Mg C ha^{-1}) while protection from grazing and shrub removal led to the greatest SOC (28.49 Mg C ha^{-1}). Declines in SOC in shrub removal treatments (with and without grazing) were compensated, in part, by enhanced aboveground C accumulation, derived mainly from woody plants. Destocking currently grazed shrublands for two decades resulted in a net C accretion, over 20 years, in the order of 6.5 Mg ha^{-1}, almost entirely through increasing belowground C. At the current price for C in Australia, the economic benefit for C accumulation from removing livestock grazing would be similar to the economic benefit of grazing. The results suggest that C farming in this semi-arid woodland system may offer an economically viable alternative management strategy to grazing, although uncertainties in future climate, C credit value, and assessment protocols present hurdles for implementing alternative management aimed at C farming.

1. Introduction

Humans have had a substantial impact on the physical and biological features of Earth, with approximately 83% of all terrestrial environments experiencing some kind of direct human influence, including urban settlement, agricultural or pastoral land use, and transport networks (Sanderson et al., 2002), resulting in various degrees of ecosystem degradation. One of the most severely impacted ecosystems is arid and semi-arid environments ('drylands'), which cover about 40% of the globe and support 40% of its human population (Maestre et al., 2012). Although disturbance (e.g., fire) has been a natural part of many dryland systems, these systems are still prone to human-induced disturbance processes, largely brought about by the introduction of domestic livestock grazing, agricultural practices, and changes to natural fire regimes (Van Auken, 2009).

One globally widespread form of land cover change in drylands is the increase in the density of woody plants, which leads to dramatic declines in pastoral productivity in many systems (e.g., Oba et al., 2000; Van Auken, 2009). The apparent mechanisms behind woody encroachment range in spatial scale from local-to-regional (e.g., increases in grazing and reductions in fire frequency) and regional-to-global (e.g., changes in N deposition, atmospheric CO_2 concentration, and climate; Sankaran and Anderson, 2009). Long periods of heavy grazing suppress grass production, and the resulting decline in grass fuel loads can alter the frequency and severity of wildfire, which exerts a controlling feedback effect on shrubs (D'Odorico et al., 2012). Increasing atmospheric CO_2 may favour the

* Corresponding author. Tel.: +61 2 9385 2194; fax: +61 2 9385 1558.
E-mail addresses: s.daryanto@student.unsw.edu.au (S. Daryanto), d.eldridge@unsw.edu.au (D.J. Eldridge), throop@nmsu.edu (H.L. Throop).
[1] Tel.: +1 575 646 5970.

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establishment of C_3 shrubs at the expense of C_4 grasses (Polley et al., 1997), and deep-rooted C_3 shrubs may be more physiologically active in dry conditions than shallow-rooted C_4 grasses (Throop et al., 2012b). Furthermore, biological feedbacks may promote the dominance of shrubs following their establishment. Enhanced soil faunal activity and deposition of nutrients beneath shrubs leads to positive plant–soil feedbacks whereby shrub growth is promoted in the interspaces at the expense of herbaceous plants (Holmgren, 2009). With generally greater levels of erosion in the interspaces and the collapse of facilitatory mechanisms due to excessive herbivory (Pugnaire et al., 2011), grass seedling establishment is infrequent. These 'novel systems' (Bridgewater et al., 2011), characterised by a greater cover and density of woody plants, are more strongly resistant to human-induced disturbances than the original systems from which they were derived (Standish et al., 2009).

The structure and composition of Australian vegetation has changed substantially in the two short centuries since European settlement. Large areas of Australia's grassland and woodland matrix have become encroached by shrubs, resulting in substantial reductions in pastoral productivity (Noble, 1997), and encouraging many pastoralists to control shrubs chemically and mechanically (Harland, 1993). One popular mechanical method is blade-ploughing (*aka* root ploughing), which has been widely promoted by government agencies to control a range of woody species (e.g., *Dodonaea*, *Senna* and *Eremophila* spp.; Robson, 1995). However, this method has been largely unsuccessful for long-term shrub eradication, and the failure of small-scale shrub removal has been blamed on regional-to-global factors such as highly variable regional rainfall (Brown et al., 1997) and rising atmospheric CO_2 concentrations (Morgan et al., 2007). As an alternative to mechanical removal, attempts to convert shrublands to their original open woodlands structure by destocking have produced inconsistent results (Angassa and Oba, 2007; Trodd and Dougill, 1998).

While the encroachment of woody plants is regarded by many as symptomatic of a degraded ecosystem (Grover and Musick, 1990), this view is mainly perpetuated in the grazing context. Considerable attention has been paid to the consequences of short- and long-term shrub removal on primary productivity (e.g., Daryanto and Eldridge, 2010; Robson, 1995). The ecological and economic consequences of shrub encroachment have received little research emphasis beyond pastoral impacts, despite evidence suggesting that encroachment can have positive impacts on multiple ecosystem functions (Eldridge et al., 2011; Maestre et al., 2009) or ecosystem 'multifunctionality' (Maestre et al., 2012). An emerging view is that encroaching shrubs, whether occurring in isolated patches or as extensive shrublands, can cause a state transition from historical grass dominance to an alternative, but stable, state (Standish et al., 2009). However, ecosystem multifunctionality in this altered stable state may still remain high (D'Odorico et al., 2012), even where herbaceous productivity is sparse or patchy. For example, shrubs may moderate surface micro-climate, reduce nutrient loss from erosional processes, enhance nutrient inputs via N fixation (Lajtha and Schlesinger, 1986), increase soil structure, stability, and the infiltration of water (Howard et al., 2012), and provide essential habitat for a range of shrub-obligate taxa (Daryanto and Eldridge, 2012; Eldridge et al., 2011; Howard et al., 2012; Maestre et al., 2009). Shrubs can also hasten the recovery of degraded systems by facilitating seedling establishment (Padilla and Pugnaire, 2006) due to their ability to moderate the effects of drought, and extremes of fire, salinity and frost (Booth et al., 1996; Richmond and Chinnock, 1994).

Carbon (C) uptake and storage may be substantially enhanced by shrub encroachment (Barger et al., 2011), and the potential ecological, economic, and atmospheric consequences of substantial shrub C uptake make this a key response variable. In North America, woody encroachment has been identified as one of the largest, albeit highly uncertain, components of the terrestrial C sink (Houghton, 2003; Pacala et al., 2001). This accumulation of C appears to be a function of enhanced below- and aboveground net primary productivity (NPP), low decomposition rates below shrubs, biochemical recalcitrance of woody litter, and organic matter stabilisation in protected soil aggregates (e.g., Knapp et al., 2008; Liao et al., 2006; Throop and Archer, 2007). Recent modelling using remote sensing data and measurements of C flux and vegetation growth as a function of climatic and soil data has identified a similar potential for C sequestration by shrubs in semi-arid system in Australian (e.g., Dean, 2011; Fensham and Guymer, 2009; Harper et al., 2007; Howden et al., 2001; Witt et al., 2011) and Mediterranean woodlands (e.g., Ruiz-Peinado et al., 2013). The magnitude of this C sink, however, is largely unknown. Also critical, but unknown, is the potential efflux of C that arises when existing shrubs are removed through management practices. This lack of information is surprising, given the large financial investment by regional and local governments in Australia in supporting the removal of shrubs from encroached woodlands on the expectation of enhanced pastoral (grazing) productivity (Robson, 1995). Recent legislation on a C price in Australia, financial incentives for landholders to sell or to lease their land for long-term C-sequestration schemes are likely (Prowse and Brook, 2011). Although shrublands typically sequester less C and are therefore less profitable than monocultural C plantations, a price on C should increase the development of 'biodiversity planting' (Prowse and Brook, 2011) or 'ecological carbon planting' through market-driven processes (Crossman et al., 2011). For Australian drylands, this ecological C planting may also provide a biodiversity and conservation co-benefit (Watson et al., 2011) and provide other ecosystem functions such as reduced salinisation and erosion (Scanlan et al., 1992).

Here we report on a study aimed at evaluating the long-term impact of different land management practices on above- and belowground C pools. In doing so, we re-evaluate the current paradigm that shrub encroached woodlands have low ecological and economic values. We do this by quantitatively assessing changes in plant and soil C stocks at different microsites in a shrub-encroached semi-arid woodland in eastern Australia subjected to varying combinations of grazing and mechanical shrub removal by blade-ploughing. We anticipated that the unploughed site from which grazing has been excluded for almost two decades would have greater total C than sites that were either ploughed or grazed, as both types of disturbance would decrease landscape-level pools of soil C. We expected therefore that the least-disturbed site would be valued more highly in terms of the ecosystem service of storing C; a service that has until recently not garnered much economic value.

2. Methods

2.1. Study area

The study was conducted at 'Wapweelah', an extensive grazing property about 35 km west of Enngonia near Bourke in north-western New South Wales, Australia (29°16′S, 145°26′E). The site falls within Gumbalie Land System (Walker, 1991), and is characterised by sandplain with low west–east trending sandy rises and dunes of Quaternary aeolian alluvium, and subject to moderate windsheeting and watersheeting. The slope is <1%, soil texture is sandy loam to loamy and the mean annual rainfall is about 312 mm, with 45% more rain falling during summer than winter (Robson, 1995).

The vegetation in the area is typical of areas in eastern Australia targeted for shrub removal by blade-ploughing (Harland, 1993).

The community is characterised by an open to dense woodland of mulga (*Acacia aneura*), which has a structure ranging from discrete islands within large patches free of woody cover to continuous groves (Tongway and Ludwig, 1990). Additionally, our site contained scattered ironwood (*Acacia excelsa*), rosewood (*Alectryon oleifolius*), leopardwood (*Flindersia maculosa*), supplejack (*Ventilago viminalis*), wilga (*Geijera parviflora*), gidgee (*Acacia cambagei*) and brigalow (*Acacia harpophylla*). The understorey layer was sparse at the time of soil sample collection, and consisted of wollybutt (*Eragrostis eriopoda*), mitchell grass (*Thyridolepsis mitchelliana*), mulga oats (*Monochather paradoxa*), rough speargrass (*Austrostipa scabra*) and other grasses and forbs. This community supports dense stands of shrubs of inverted-cone morphology (Y-shaped) such as turpentine (*Eremophila sturtii*), narrow-leaf hopbush (*Dodonaea viscosa* var. *angutissima*), budda (*Eremophila mitchellii*) and hemispherical shrubs such as green turkey bush (*Eremophila gilesii*) and silver cassia (*Senna artemisiodes*; Robson, 1995; Walker, 1991). Shrub morphology is known to affect the capacity of shrubs to retain litter beneath their canopies, with hemispherical-shaped shrubs retaining a greater amount of litter beneath their canopies than shrubs with an inverted-cone morphology (*aka* inverted-cone shrubs; De Soyza et al., 1997). This morphological difference could influence soil carbon and nitrogen pools.

2.2. Shrub removal treatments

In 1990, three replicate blocks (1–5 km apart) of 400 m × 200 m were established and each was divided into four plots of 100 m wide by 200 m long. The two central plots were fenced in a 6 m high, herbivore-proof fence while the two outer plots were subject to grazing by sheep and cattle, with typical stocking rates ranging from 0.07 sheep ha^{-1} (i.e., a non-lactating ewe) during dry periods to 0.5 sheep ha^{-1} during wet periods (Robson, 1995). Kangaroos, feral goats, and feral European rabbits also grazed the outer plots, but at unknown densities. One of the two central plots was left unploughed and the other one was ploughed. Similarly, one of the outer (grazed) plots was ploughed and the other one left unploughed, which resulted in four treatments based on all possible grazing by ploughing combinations (Table 1). These four treatments represent different land management scenarios reflecting the range of land management practices in shrub-encroached woodlands in eastern Australia. The 'conventional pastoral' treatment represents the most common land management scenario in eastern Australia and is based on set stocking with no shrub removal, while 'active pastoral' indicates shrub removal combined with various levels of set stocking in order to recoup some of the costs associated with shrub removal. 'Conventional conservation' is a strategy practiced within national parks and conservation reserves whereby livestock are removed but shrubs are not controlled. The 'active conservation' strategy, however, is rarely used, and represents an experimental treatment (Robson, 1995) that may have some management value within a conservation context such as the removal of isolated shrubs around fences or other infrastructure, or shrub removal in the absence of grazing to meet specific conservation objectives such as creating habitat for key plant or animal species (Table 1).

Shrubs were removed with a single pass of a 4.2 m wide single-tyned 'Stationmaster' blade-plough pulled by a 90 kW crawler tractor. Only shrubs were targeted during ploughing and cutting depth was maintained at 20–30 cm to sever the taproots of most shrubs, particularly *Eremophila* spp., to prevent resprouting (Wiedemann and Kelly, 2001). Examination of fallen shrubs confirmed a satisfactory ploughing effect over most of the treated area (Robson, 1995). The remains of shrubs, either those pushed over by ploughing or those that died naturally, were thus considered as coarse woody debris (CWD or log mounds; Tongway et al., 1989). They act as a barrier against the movement of runoff and sediment and become sites of soil and organic matter accumulation. They often form fertile mounds (up to 10 m^2 area) with higher available nutrients and small animal activity (Daryanto and Eldridge, 2012; Daryanto et al., 2012).

2.3. Field sampling and laboratory analyses

In September 2010, two decades after ploughing and fencing, we established three 50 m long by 2 m wide belt transects through the centre of each of the 12 plots to measure the cover of different landscape elements. We identified six landscape elements across the study site: (1) N-fixing trees (i.e., *Acacia* spp.), (2) non N-fixing trees (e.g., *G. parviflora*), (3) inverted-cone shaped shrubs (e.g., *E. sturtii*), (4) hemispherical shrubs (e.g., *Senna artemisioides*), (5) mounds of CWD, and (6) interspaces. For each landscape element on each plot, we collected between two and five intact soil cores (4.2 cm diameter, 30 cm depth); the number of cores increased with the relative cover of each landscape element. Cores were separated into four depth increments (0–5, 5–10, 10–20, and 20–30 cm). For each of the 12 plots, we combined all depth increments obtained from within each of the six landscape elements in order to obtain one bulked sample for each landscape element per plot for each depth. Soil cores for shrubs and trees were taken at mid-canopy positions, while interspace soils were sampled on bare soil surfaces at least 5 m from any other landscape elements. For CWD mounds, cores were taken about 10 cm upslope of the mounds. Soil cores were air dried, passed through a 2 mm sieve to remove any roots or organic debris, ground, and subjected to combustion analysis (LECO CNS-2000; LECO Corporation, St. Joseph, MI, USA) in order to quantify soil organic C (SOC; i.e., without carbonates) and N concentrations on a per mass basis (i.e., %SOC and soil %N, respectively). Additional cores for each soil depth by landscape element combination were collected to determine bulk density and to allow us to convert %SOC and %N to area-based pools. Bulk density (Mg m^{-3}) was calculated as the mass of the fine earth (<2 mm) fraction divided by the volume of the entire core to avoid overestimating the mass of fine earth when stones were present (Throop et al., 2012a).

Table 1
Summary of the four grazing by ploughing combinations and their relative extent in eastern Australia.

Scenario	Description	Extent	Disturbance intensity
Conventional conservation	Historically grazed, then ungrazed for 20 years prior to sampling	Localised (<10% of landholders)	Low
Active conservation	Historically grazed, one-off ploughing, then ungrazed for 20 years prior to sampling	Very uncommon (<1% of landholders)	Moderate
Conventional pastoral	Historically and continuosly grazed for 20 years prior to sampling	Widespread (>50% of landholders)	Moderate
Active pastoral	Historically grazed, followed by a one-off ploughing, then continued to be grazed for 20 years prior to sampling	Very common (25–50% of landholders)	High

Derived from Daryanto and Eldridge (2010).

2.4. Landscape element and plot-level carbon estimates

Stocks of SOC and soil N on a per area basis were calculated as a product of C or N concentration (%SOC or soil %N), bulk density, and soil depth. This was done firstly by calculating the segment delimited by polynomial equations ($R^2 > 0.999$; $n = 72$) that specifically describe the relationship between %SOC or soil %N of each landscape element in each of the 12 plots and its corresponding depths using Matlab R2010b. Depth-specific bulk density was then used to convert the %SOC and soil %N to g SOC m^{-2} or soil N m^{-2} for each depth, enabling comparison of SOC or soil N across landscape elements on an areal basis (hereafter 'landscape element SOC' or 'landscape element soil N'; Harms et al., 2005). Mass of SOC or soil N for all landscape elements within a management treatment plot (hereafter 'plot-level SOC' or 'plot-level soil N') was calculated by multiplying landscape element SOC or landscape element soil N by the percent cover of the corresponding landscape element.

We estimated the mass of woody plant tissue in each landscape element based on allometric equations and area-adjusted cover of woody plants and CWD in each plot. Percent cover data were obtained from belt transects (see Section 2.3); plants were included if their stem was rooted within the belt transect. For the upperstorey, we measured the height and canopy width of all trees and shrubs, except for tall (>2 m high) *A. aneura*, for which we used diameter at breast height (DBH). Biomass of individual woody plants was calculated using species-specific algorithms for *A. aneura* woodlands (Suganuma et al., 2006) based on plant width and height or DBH. Root biomass was estimated as 27.14% of aboveground biomass, except for *A. aneura* (40.2%), based on previously established relationships (Suganuma et al., 2006). Total mass of CWD was estimated using a 'standard log' (Andrew et al., 1979) whose weight and dimensions were used as a comparison against all CWD found in all plots. A further 12 logs were estimated and their mass assessed across the 12 sites to derive predictive equations similar to the groundstorey biomass. We included all CWD where 50% or more of the material was within the belt transect. Mass of C was estimated from all aboveground (upperstorey and understorey), root, and CWD biomass by assuming 50% C content (Snowdon et al., 2000).

Total aboveground herbaceous biomass was estimated at the plot level within ten to twenty five 0.25 m^2 (0.5 m × 0.5 m) quadrats per plot by an experienced observer (David Eldridge). A greater number of quadrats was sampled in plots with more herbaceous cover. Biomass was estimated using a portfolio of photographic standards of known weight, which are used widely in rangeland assessment (Eldridge and Koen, 2003). At the completion of measurements we estimated the biomass within an additional five quadrats for each plot, where all aboveground material was removed, dried at 60 °C for 72 h and weighed. We then established relationships between actual mass of clipped material and estimates based on the photographic standards in order to correct our field-based estimates. The predictive power of these regressions was relatively strong ($R^2 = 0.74$–0.89; $P < 0.001$, $n = 60$).

Total C pools were estimated for each of the 12 management treatment plots (Fig. 1). Plot-level belowground C was the sum of plot-level SOC and plot-level root C while plot-level aboveground C was the sum of plot-level aboveground woody C (shrub aboveground C, tree aboveground C, and CWD) and plot-level aboveground herbaceous C. The sum of plot-level aboveground and belowground C was defined as plot-level total C.

2.5. Statistical analyses

Univariate analysis using a randomised-block ANOVA was used to determine the effect of treatments on the percentage cover of different landscape elements while differences in %SOC, soil %N, C:N ratio, and bulk density in relation to treatment and landscape elements were analysed using a mixed-models ANOVA. The main plot of the mixed-model ANOVA considered block ($n = 3$) and treatment ($n = 4$) effects, and the sub-plot landscape element effects (i.e., interspace, CWD mound, inverted-cone shrub, hemispherical shrub, N-fixing tree, non N-fixing tree) and their interactions with treatment. Univariate analyses for assessing differences in plot-level soil N, aboveground, belowground and total C among treatments were tested using randomised-block ANOVA after scaling the data up to the plot level by adjusting for the percent cover of different landscape units at each plot. Data were checked for normality and homogeneity of variance (Levene's test) using Minitab

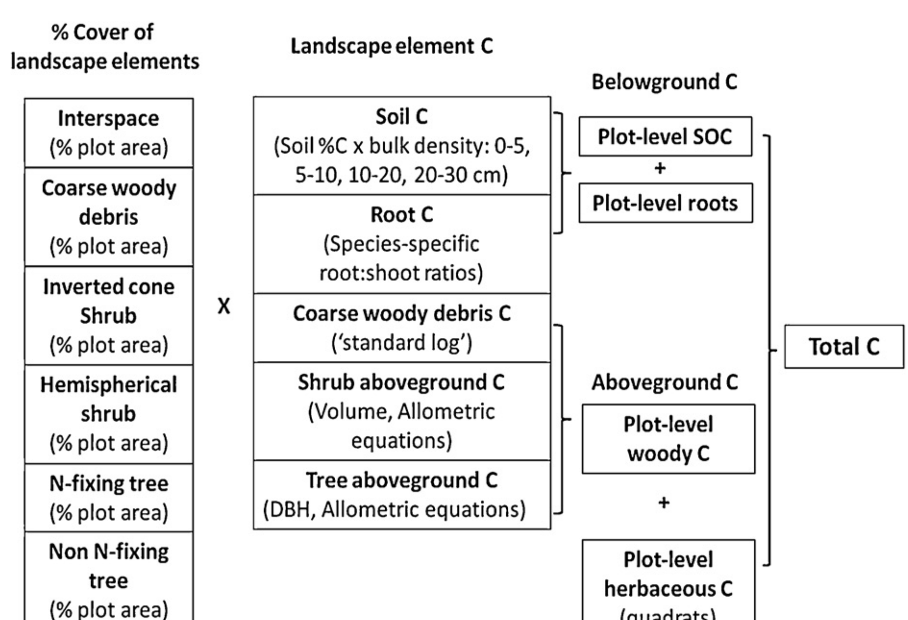

Fig. 1. Summary of the components and methods used for constructing C pools. Plot-level C pools were calculated as the sum of all measured aboveground and belowground components. Except for herbaceous aboveground C, landscape element C pools were multiplied by the corresponding percent area of each landscape element to obtain plot-level C pools for each management treatment.

Table 2
Mean (±SE) of cover (%) of the six landscape elements under the four different management treatments.

Landscape element	Conventional conservation		Active conservation		Conventional pastoral		Active pastoral	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Interspace	65.2	5.0	62.9	2.1	69.4	3.2	72.2	3.2
Coarse woody debris	6.0	2.2	4.6	1.7	6.7	3.0	5.0	1.6
Inverted cone shrub	12.0	4.6	15.5	4.0	13.4	6.9	11.0	2.8
Hemispherical shrub	8.0	2.9	4.4	1.5	6.5	5.0	6.0	3.7
N-fixing tree	6.6[ab]	1.4	8.9[b]	3.2	1.5[a]	0.3	2.8[a]	1.6
Non N-fixing tree	2.3	0.3	3.7	0.7	3.0	1.2	1.2	0.4

Significant differences in cover of landscape elements among treatments at $P<0.05$ are denoted by different letter superscripts. Significant differences were apparent only for N-fixing trees.

15 prior to analyses. *Post hoc* differences in means were tested using Least Significant Difference testing.

3. Results

3.1. Distribution of landscape elements in management treatment plots

Averaged across all management treatments, interspaces occupied 67 ± 0.6% (mean ± SE) of the surface area, inverted cone-shaped shrubs 13 ± 0.9%, and the other landscape elements each occupied 3–6% of the area (Table 2). With the exception of N-fixing trees, there were no differences in the cover of landscape elements among the four management types. The cover of N-fixing trees (e.g., *A. aneura* and *A. excelsa*) was significantly greater in the active conservation plots compared with those under active pastoral and conventional pastoral treatment ($F_{3,6}=12.32$, $P=0.013$; Table 2).

3.2. Carbon and nitrogen in landscape elements

Averaged across all depths and management treatments, soils in the interspaces had the highest bulk density ($F_{5,40}=3.39$, $P=0.012$), but a consistently lower %SOC ($F_{5,40}=13.09$, $P<0.001$), soil %N ($F_{5,40}=11.62$, $P<0.001$), and C:N ratio ($F_{5,40}=8.16$, $P<0.001$; Table 3) than the other landscape elements. There were also significant treatment by landscape element interactions for %SOC ($F_{15,40}=3.17$, $P=0.002$) and soil %N ($F_{15,40}=3.15$, $P=0.002$), with levels of both %SOC and soil %N similar under all elements for all treatments, except for the conventional pastoral treatment where values under non N-fixing trees were substantially greater (Appendix A). When analysed by depth, the same trends for %SOC and soil %N were also observed for the three topmost layers (0–5, 5–10, and 10–20 cm) but not in the deepest soil layer (20–30 cm; $P=0.093$ and $P=0.089$ for %SOC and soil %N, respectively). In contrast, bulk density patterns were driven by changes in the surface soils, as surface soil (0–5 cm) drove the pattern of greatest bulk density in interspace soils compared to other landscape elements ($F_{5,40}=3.60$, $P=0.009$). This pattern was consistent across treatments (Tables 4 and 5).

3.3. Carbon and nitrogen pools at the plot level

At the plot level, grazing and ploughing caused significant declines in belowground C (soil + root C; $F_{3,6}=11.39$, $P=0.007$). The mass of plot-level belowground C on the least disturbed treatment (Conventional Conservation; 30.1 Mg ha^{-1}) was almost 50% greater than that at sites under the most disturbed treatment (Active Pastoral; 19.1 Mg ha^{-1}; Fig. 2A). Most of this difference was due to storage in the uppermost layer (0–5 cm) of the soil profile ($F_{3,6}=45.70$, $P<0.001$; Table 4), with an almost five-fold greater SOC at 0–5 cm in the Conventional Conservation (16.3 Mg ha^{-1}) than the Active Pastoral (3.3 Mg ha^{-1}) plots. The interspaces contributed most to the SOC in the 0–5 cm soil layer (Table 4), largely due to their areal (Table 2) and bulk density extent (Table 3). We also detected some significant differences in the SOC in deeper soil layers, with soils under Active Conservation consistently having the highest SOC in the 5–10 cm layer (4.9 Mg ha^{-1}; $F_{3,6}=5.83$, $P=0.03$) and 10–20 cm layer (6.2 Mg ha^{-1}; $F_{3,6}=9.05$, $P=0.01$) compared with the other treatments. These substantial differences in SOC storage in the upper layers, however, attenuated strongly with depth (20–30 cm; $P=0.07$). We did not detect any differences, however, in soil N with different treatments within each corresponding soil depth (Table 5).

In contrast to belowground C, there was a trend towards increased aboveground biomass in the ploughed treatments, although this pattern was not significant ($P=0.13$; Table 6). Total C was generally lowest in the conventional pastoral treatment. However, there were no significant differences among

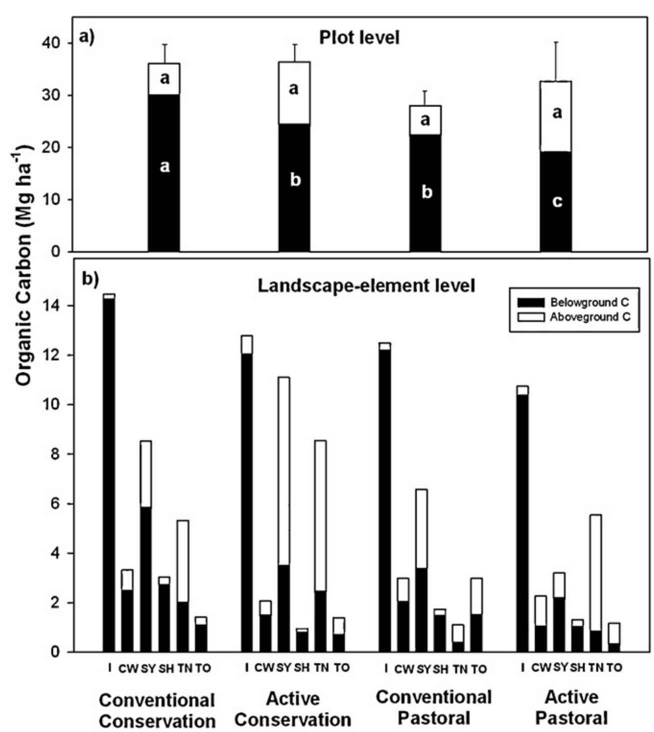

Fig. 2. (A) Mean mass of aboveground, belowground (soil + root), and mean mass (±SE) of total (aboveground + belowground) C (Mg ha^{-1}) for different land management treatments and (B) Mean mass of belowground and aboveground C, subdivided into landscape elements within management treatments. Values represent the product of aerial cover of each landscape element and the area-based C concentration. Landscape elements are abbreviated as I = interspace, CW = coarse woody debris, SY = inverted-cone shrub, SH = hemispherical shrub, TN = N-fixing tree, TO = Non N-fixing tree. Different letters indicate a significant difference at $P<0.05$. Letters in Fig. 1A indicate that difference is only significant for belowground C, but not aboveground and total C.

Table 3
Mean (±SE) of %SOC, soil %N, C:N ratio, and bulk density averaged across soil depth and the four land management treatments for soil under different landscape elements.

Attribute	Interspace		CWD mound		Inverted cone-shaped shrub		Hemispherical shrub		N-fixing tree		Non N-fixing tree	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Soil organic C (%)	0.32[a]	0.03	0.59[b]	0.06	0.52[b]	0.32	0.52[b]	0.05	0.63[b]	0.06	0.78[c]	0.96
Soil N (%)	0.038[a]	0.002	0.049[b]	0.004	0.050[b]	0.002	0.050[b]	0.003	0.057[c]	0.005	0.066[d]	0.007
C:N ratio	8.38[a]	0.50	12.04[c]	0.66	10.22[b]	0.38	10.19[b]	0.73	10.63[b]	0.55	11.33[bc]	0.45
Bulk density (Mg m^{-3})	1.77[a]	0.04	1.68[ab]	0.06	1.69[ab]	0.05	1.64[b]	0.04	1.63[b]	0.04	1.60[b]	0.06

SE, standard error of the mean. Different superscripts indicate a significant difference in values among different landscape elements at $P < 0.05$.

management treatments ($P = 0.18$; Fig. 2A), reflecting the combination of disturbance-induced declines in belowground C and positive aboveground responses to ploughing. The contribution of aboveground C to total C was greater for treatments that were ploughed (Active Conservation = 32.6%; Active Pastoral = 41.6%) than for unploughed (Conventional Conservation = 16.6%; Conventional Pastoral = 20.1%; Fig. 2A and B).

4. Discussion

Large areas of semi-arid woodland in eastern Australia have been encroached by woody plants (Noble, 1997), presenting substantial problems for land managers whose primary enterprise has been the grazing of sheep and cattle on native pastures. Many pastoral managers have considered few alternative land uses for shrub-affected woodlands apart from opportunistic harvesting of feral goats and, in some cases, ecotourism (Prowse and Brook, 2011). Our study sought to test whether C farming could be an alternative (i.e., profitable) land use option for shrub-affected woodlands. Increasing disturbance resulted in a 37% decline (from 30 to 19 Mg ha^{-1}) in belowground C from the least disturbed (Conventional Conservation) to the most disturbed (Active Pastoral) treatments. These values are remarkably close to those from studies conducted by Harms et al. (2005) on the clearing of mulga country in Queensland using similar methods. The observed drop in SOC was driven by the top 5 cm of soil, where SOC was five-times greater under the least disturbed treatment (16.3 Mg ha^{-1}) than under the most disturbed treatment (3.3 Mg ha^{-1}; Table 4). While %SOC and soil %N were least in the unvegetated interspaces (Table 3), the extensive cover of interspaces across all treatments (Table 2) resulted in them supporting the largest pools of both SOC (Table 4) and soil N (Table 5). In contrast to belowground C, there was a trend towards increasing aboveground C in the ploughed treatments. Overall, our results indicate that surface disturbances such as grazing, and physical removal of shrubs by ploughing, lead to reductions in SOC, but encroachment by increasing the cover and biomass of woody plants partially compensates for this reduction. The importance of shrubs thus increases with disturbance since SOC after land clearing was also associated with vegetation type (i.e., root distribution; Harms et al., 2005). *Acacia* spp., in particular, have denser roots in the surface soils than other species (Burrows, 1976; Suganuma et al., 2006) which may explain why maintaining native vegetation is important if C sequestration is a management objective.

4.1. Depth effects on carbon and nitrogen

Observed declines in SOC with depth are consistent with global observations of soil C and nutrient accumulation in the topsoil and their rapid attenuation with depth (e.g., Jobbágy and Jackson, 2001). Responses to disturbance also attenuated with depth, such that a reduction in SOC pools was not apparent at depths > 5 cm even though the depth of ploughing was substantially greater (~30 cm deep; Robson, 1995). While the relative contribution of SOC in the 0–5 cm layer declined from Conventional to Active Conservation, i.e., with only the imposition of ploughing, further disturbances by grazing did not lead to additional declines in SOC pools (Table 4).

Trends in N pools, however, did not reflect those for C, with no apparent difference in N pool distribution with depth among treatments or landscape elements (Table 5). Part of the increased %SOC but not soil %N could have been due to the difference in the composition of biological soil crusts; complex associations between the surface soil layers and microphytic organisms (e.g., cyanobacteria, lichens, mosses; Kidron et al., 2010). Crusts are known to fix C and N as well as moderating their loss from the ecosystem (Barger et al., 2006). They differ, however, in their N fixing capability with darker (i.e., increased chlorophyll, protein and carbohydrates) crusts able to fix more N than the light-coloured ones (Kidron et al., 2010). The former type of crusts, however, only occurred in stable, undisturbed soils (Kidron et al., 2009). Because of the strong suppressive effect of disturbance on biological soil crust cover (e.g., trampling and ploughing) and their long time to recovery (e.g., Daryanto and Eldridge, 2010), it was likely that the dominant biological crusts at our sites were those that had less capability to fix N.

4.2. Soil and aboveground carbon among different landscape elements

While interspaces contained the largest SOC pools (due to large area and high bulk density), this large pool of SOC was highly responsive to disturbance by grazing or ploughing. In contrast, differences in %SOC and soil %N increased markedly from the interspaces, through coarse woody debris and shrubs, to trees, with concentrations under large, non N-fixing trees twice that in the interspaces (Table 3). Our results are consistent with the large body of empirical evidence indicating the accumulation of essential resources (nutrients, water, litter) under shrubs and coarse woody debris (e.g., Daryanto et al., 2012; Tongway and Ludwig, 1996; Wang et al., 2007). Examination of C pools in relation to landscape elements showed that inverted cone-shaped shrubs contributed relatively large amounts of C at the undisturbed sites (~6 Mg ha^{-1}; Table 4). We attribute this to increased %SOC under shrubs, as the cover of these cone-shaped shrubs was consistent across treatments (11–16%; Table 2) and soil bulk density was generally lower around these shrub hummocks (Table 3).

Our results suggest that the importance of vegetation patches relative to interspaces as pools of C increases as sites become degraded. Much of this shift is driven, however, by C in aboveground biomass. While ploughing generated SOC loss, it tended to increase aboveground C due to vegetation shift to a younger, more productive age state, similar to patterns reported by Hughes et al. (2006). Therefore, unlike belowground C pools, aboveground pools were lowest in the interspaces (i.e., herbaceous cover) and for hemispherical-shaped shrubs, and greatest for N-fixing or inverted cone-shaped shrubs (Table 6). Ploughing is known to stimulate shrub (e.g., *Eremophila* spp.) regrowth, either from seed or belowground woody structures such as root suckering or epicormic buds (Chinnock, 2007; Wiedemann and Kelly, 2001).

Table 4
Mean (±SE) of bulk density, %SOC and SOC across the four land management treatments for soil under different landscape elements.

Landscape element	Conventional conservation						Active conservation						Conventional pastoral						Active pastoral					
	Bulk density ($Mg\,m^{-3}$)		SOC (%)		SOC ($Mg\,ha^{-1}$)		Bulk density ($Mg\,m^{-3}$)		SOC (%)		SOC ($Mg\,ha^{-1}$)		Bulk density ($Mg\,m^{-3}$)		SOC (%)		SOC ($Mg\,ha^{-1}$)		Bulk density ($Mg\,m^{-3}$)		SOC (%)		SOC ($Mg\,ha^{-1}$)	
	χ	SE	χ	SE	χ	SE	χ	SE	χ	SE	χ	SE	χ	SE	χ	SE	χ	SE	χ	SE	χ	SE	χ	SE
0–5 cm																								
I	1.90	0.103	0.32	0.027	8.13	0.353	1.49	0.126	0.56	0.118	2.66	0.679	1.66	0.168	0.53	0.174	2.95	0.703	1.63	0.038	0.41	0.117	2.32	0.508
CW	1.85	0.077	0.73	0.113	1.44	0.444	1.34	0.117	0.85	0.127	0.26	0.099	1.26	0.193	1.17	0.143	0.44	0.138	1.65	0.214	0.85	0.221	0.30	0.066
SY	1.46	0.063	0.84	0.122	3.33	1.293	1.48	0.036	0.69	0.104	0.83	0.309	1.43	0.092	0.90	0.151	0.88	0.434	1.50	0.099	0.75	0.138	0.64	0.205
SH	1.70	0.199	0.65	0.063	1.64	0.574	1.50	0.078	0.63	0.033	0.20	0.054	1.46	0.168	1.19	0.261	0.41	0.275	1.34	0.188	0.71	0.097	0.26	0.162
TN	1.53	0.008	0.56	0.055	1.16	0.302	1.44	0.152	1.05	0.157	0.63	0.203	1.48	0.090	1.17	0.257	0.13	0.029	1.51	0.133	1.23	0.149	0.25	0.123
TO	1.48	0.050	0.99	0.110	0.64	0.110	1.23	0.304	0.81	0.051	0.19	0.065	1.25	0.074	2.15	0.134	0.41	0.163	1.47	0.059	1.27	0.267	0.10	0.030
5–10 cm																								
I	1.94	0.025	0.27	0.030	1.67	0.094	1.98	0.284	0.52	0.090	3.11	0.751	1.82	0.207	0.33	0.056	2.12	0.307	1.79	0.010	0.30	0.066	1.86	0.230
CW	1.93	0.133	0.49	0.051	0.30	0.096	1.97	0.168	0.59	0.118	0.28	0.127	1.44	0.089	0.89	0.188	0.40	0.183	1.72	0.076	0.58	0.161	0.21	0.031
SY	1.89	0.119	0.68	0.089	0.80	0.365	1.94	0.147	0.49	0.042	0.76	0.240	1.82	0.210	0.60	0.079	0.73	0.357	1.68	0.152	0.47	0.047	0.43	0.113
SH	1.75	0.119	0.41	0.064	0.34	0.130	1.64	0.117	0.48	0.013	0.17	0.046	1.95	0.198	0.63	0.106	0.35	0.143	1.71	0.038	0.46	0.106	0.22	0.105
TN	1.92	0.101	0.35	0.039	0.25	0.057	1.50	0.222	0.83	0.165	0.46	0.127	1.63	0.060	0.66	0.145	0.08	0.020	1.66	0.219	0.79	0.056	0.17	0.087
TO	1.73	0.079	0.67	0.058	0.13	0.024	1.53	0.342	0.56	0.059	0.16	0.044	1.61	0.105	1.16	0.177	0.31	0.120	1.78	0.034	0.79	0.182	0.08	0.027
10–20 cm																								
I	1.83	0.069	0.20	0.010	2.48	0.184	1.73	0.066	0.31	0.058	3.57	0.644	1.71	0.097	0.30	0.032	3.46	0.304	1.73	0.142	0.25	0.035	2.94	0.198
CW	1.93	0.100	0.41	0.066	0.44	0.143	1.71	0.170	0.45	0.106	0.67	0.168	1.60	0.227	0.59	0.106	0.53	0.201	1.52	0.144	0.39	0.028	0.28	0.077
SY	1.74	0.177	0.47	0.045	1.01	0.449	1.65	0.175	0.36	0.018	1.03	0.417	1.64	0.165	0.42	0.087	0.92	0.461	1.64	0.076	0.32	0.021	0.60	0.153
SH	1.64	0.047	0.29	0.024	0.45	0.154	1.65	0.164	0.33	0.034	0.24	0.078	1.72	0.077	0.42	0.074	0.38	0.258	1.53	0.107	0.35	0.047	0.28	0.151
TN	1.72	0.083	0.32	0.018	0.34	0.101	1.60	0.057	0.52	0.113	0.76	0.317	1.58	0.054	0.46	0.121	0.11	0.030	1.62	0.113	0.52	0.048	0.25	0.160
TO	1.95	0.167	0.42	0.029	0.20	0.032	1.41	0.334	0.37	0.031	0.19	0.059	1.71	0.076	0.86	0.177	0.41	0.131	1.59	0.081	0.46	0.102	0.08	0.023
20–30 cm																								
I	1.68	0.064	0.18	0.011	1.98	0.146	1.74	0.044	0.25	0.049	2.70	0.475	1.77	0.060	0.29	0.027	3.67	0.574	1.98	0.293	0.25	0.021	3.27	0.325
CW	1.74	0.139	0.32	0.039	0.32	0.104	1.68	0.022	0.38	0.084	0.30	0.138	1.87	0.182	0.51	0.096	0.67	0.363	1.68	0.085	0.32	0.021	0.26	0.082
SY	1.81	0.225	0.38	0.012	0.74	0.189	1.66	0.063	0.33	0.018	0.88	0.291	1.92	0.160	0.38	0.029	0.87	0.384	1.74	0.173	0.27	0.027	0.53	0.176
SH	1.43	0.121	0.56	0.288	0.30	0.076	1.68	0.070	0.27	0.034	0.20	0.083	1.68	0.051	0.42	0.070	0.34	0.201	1.90	0.237	0.52	0.282	0.27	0.133
TN	1.67	0.064	0.24	0.020	0.27	0.063	1.70	0.070	0.43	0.100	0.62	0.268	1.63	0.070	0.42	0.092	0.10	0.021	1.88	0.301	0.43	0.020	0.19	0.092
TO	1.57	0.048	0.36	0.029	0.13	0.026	1.77	0.035	0.29	0.007	0.19	0.026	1.81	0.076	0.87	0.115	0.40	0.117	1.75	0.197	0.39	0.065	0.07	0.019

SE, standard error of the mean; I, interspaces; CW, coarse woody debris; SY, inverted-cone shrubs; SH, hemispherical shrubs; TN, N-fixing tree; TO, non N-fixing tree.

Table 5
Mean (±SE) of soil %N and soil N across the four land management treatments for soil under different landscape elements.

Landscape element	Conventional conservation				Active conservation				Conventional pastoral				Active pastoral			
	Soil N (%)		Soil N (kg ha^{-1})		Soil N (%)		Soil N (kg ha^{-1})		Soil N (%)		Soil N (kg ha^{-1})		Soil N (%)		Soil N (kg ha^{-1})	
	χ	SE	χ	SE	χ	SE	χ	SE	χ	SE	χ	SE	χ	SE	χ	SE
0–5 cm																
I	0.04	0.002	257.48	16.070	0.05	0.007	247.41	31.208	0.06	0.010	324.46	68.890	0.04	0.009	247.52	30.903
CW	0.06	0.006	29.53	8.519	0.07	0.006	21.52	7.628	0.09	0.014	39.77	17.724	0.06	0.004	23.00	7.510
SY	0.08	0.012	73.45	29.174	0.06	0.009	72.19	26.867	0.08	0.011	76.64	33.406	0.06	0.010	47.45	14.648
SH	0.06	0.003	41.28	14.631	0.06	0.006	18.62	4.993	0.10	0.013	47.94	37.163	0.06	0.008	23.14	15.611
TN	0.05	0.005	25.68	4.900	0.08	0.011	50.48	16.395	0.10	0.016	10.93	1.342	0.10	0.009	20.39	10.609
TO	0.08	0.012	14.47	2.252	0.07	0.003	15.97	5.464	0.16	0.016	33.53	14.965	0.10	0.020	7.65	2.070
5–10 cm																
I	0.03	0.002	227.80	8.377	0.04	0.004	266.55	43.809	0.04	0.006	312.86	128.435	0.04	0.005	227.22	5.400
CW	0.05	0.004	28.31	9.209	0.04	0.006	21.80	9.328	0.08	0.025	34.90	19.942	0.04	0.002	17.95	2.274
SY	0.06	0.007	70.44	32.210	0.05	0.006	72.96	26.496	0.06	0.003	62.25	29.266	0.04	0.003	36.21	8.234
SH	0.05	0.004	29.51	9.008	0.04	0.020	15.21	4.441	0.06	0.005	48.38	40.295	0.04	0.004	22.67	10.165
TN	0.04	0.003	25.60	5.142	0.06	0.016	35.07	9.064	0.06	0.007	7.54	0.940	0.07	0.006	16.48	8.959
TO	0.06	0.005	11.37	0.841	0.05	0.002	13.80	4.073	0.09	0.017	26.60	10.713	0.07	0.011	6.33	1.968
10–20 cm																
I	0.03	0.003	401.79	22.162	0.03	0.001	380.63	14.569	0.03	0.003	397.54	67.657	0.03	0.004	420.90	20.871
CW	0.04	0.003	51.40	19.461	0.04	0.006	41.36	20.948	0.06	0.012	50.24	17.549	0.03	0.001	24.36	9.061
SY	0.05	0.002	97.69	38.801	0.04	0.006	96.96	36.663	0.05	0.006	88.36	40.417	0.03	0.001	59.26	14.233
SH	0.04	0.001	43.61	17.752	0.04	0.004	27.32	9.112	0.05	0.003	67.49	25.515	0.04	0.002	33.94	21.816
TN	0.04	0.001	39.43	9.891	0.05	0.009	66.69	26.171	0.05	0.010	9.98	2.240	0.05	0.003	25.07	16.080
TO	0.04	0.003	19.00	0.416	0.04	0.002	19.87	6.534	0.07	0.002	40.73	16.213	0.04	0.005	7.28	1.786
20–30 cm																
I	0.03	0.002	376.41	20.871	0.04	0.003	399.14	33.332	0.04	0.008	501.19	151.734	0.04	0.001	531.04	111.445
CW	0.04	0.002	42.01	15.328	0.04	0.004	37.82	20.789	0.04	0.010	55.70	23.641	0.03	0.001	25.53	7.666
SY	0.04	0.004	80.33	23.410	0.04	0.002	96.08	28.366	0.06	0.008	87.65	35.298	0.03	0.001	59.92	15.506
SH	0.04	0.003	34.14	8.593	0.03	0.005	27.42	10.624	0.05	0.006	69.38	58.681	0.05	0.020	20.92	20.920
TN	0.03	0.004	37.28	7.080	0.04	0.006	60.86	23.224	0.05	0.010	11.90	0.791	0.05	0.002	20.27	9.083
TO	0.04	0.005	13.60	1.709	0.04	0.003	22.40	1.609	0.07	0.005	42.67	17.477	0.04	0.005	6.90	1.916

SE, standard error of the mean; I, interspaces; CW, coarse woody debris; SY, inverted-cone shrubs; SH, hemispherical shrubs; TN, N-fixing tree; TO, non N-fixing tree.

Our previous studies at this site show that ploughing increased the overall densities of woody plants (Daryanto and Eldridge, 2010), and hence the high carbon on these ploughed plots. Both ploughed treatments (Active Conservation and Pastoral) had a substantial density of shrubs (mainly *D. viscosa* and some *Eremophila* spp.) as well as some N-fixing trees (mainly *A. aneura*) that made up a substantial percentage (~82%) of the total aboveground C pool. As subsequent grazing would also favour shrub regeneration by removing competitors such as grasses, the unpalatable *Eremophila* spp. would tend to dominate areas with a history of heavy stock grazing (Tiver and Andrew, 1997), likely compensating in aboveground C for the loss of SOC.

4.3. Managing shrubland for ecosystem carbon stocks

These results indicate that the consequences of even one disturbance event can persist for decades in this system. In most low productivity systems, such as semi-arid rangelands, plants recover slowly after disturbance (Lunt et al., 2007). It is likely that the decline in surface soil C was due to the combination of enhanced heterotrophic respiration following the disturbance (Schulze, 2006) and limited nutrient and C input due to lack of vegetative cover. Expanding shrub biomass in the decades since ploughing suggest that the depressed soil C may decline with time, and it is likely that it has recovered, at least in part, from surface disturbance. While the active conservation scenario might yield approximately the same amount of plot-level C, with cost ±$A 45 ha^{-1} in 2009 (NSW Department of Primary Industry, 2010), the economics of ploughing will depend on many things such as the opportunity cost of not grazing (i.e., the foregone grazing value of land that is left to thicken with shrubs) and the cost of treatment (e.g., labour, machinery, and fuel). This cost, however, may vary depending on soil texture, soil type, soil moisture, plough model, tractor type, shrub size, and density (Harland, 1993).

Table 6
Mean (±SE) of aboveground C (Mg ha^{-1}) of the landscape elements under the four different management treatments.

Landscape element	Conventional conservation		Active conservation		Conventional pastoral		Active pastoral	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Coarse woody debris	0.82	0.154	0.56	0.071	0.97	0.361	1.21	0.213
Inverted cone shrub	2.67	0.291	7.60	2.440	3.17	1.360	10.06	8.050
Hemispherical shrub	0.30	0.022	0.14	0.099	0.26	0.142	0.28	0.117
N-fixing tree	3.31	1.500	6.07	2.780	0.68	0.157	4.68	4.030
Non N-fixing tree	0.31	0.004	0.66	0.004	1.45	0.800	0.84	0.197

Interspaces were not included as all biomass was herbaceous and assessed at the plot level. There were no differences in mass of C of any landscape element among treatments at $P < 0.05$.

Our one-off treatments were conservative; under more active pastoral management, follow-up treatment such as spot spraying with herbicide, active grazing with goats or burning with a hand-held burner would be applied a few years after any signs of shrub regrowth following ploughing (Noble et al., 2005; Vitelli and Madigan, 2004). Ploughed treatments such as ours might have even been re-ploughed, depending on the success of the original treatment, resulting in further surface destabilisation and removal of C.

Removal of domestic and feral herbivores for two decades coupled with no soil disturbance produced the greatest C accumulation benefits, consistent with observations from other exclosure studies in eastern Australian woodlands (Witt et al., 2011). Our results suggests a net C accretion in the order of 6.5 Mg ha^{-1} of total C when management changes from a conventional pastoral system to one based on conservation with no active soil disturbance (Conventional Conservation), almost entirely through increasing belowground C (Fig. 2A). While we have no data on annual changes over the past two decades, this potentially greater C pool represents a net financial benefit of about \$A 548 ha^{-1} over that time (\$A 27.40 ha^{-1} y^{-1}), based on the Australian Government's current price for carbon of \$A 23 Mg^{-1} of CO_2 equivalents (=\$A 84.27 Mg^{-1} of C; Australian Government, 2012; Dean, 2011). Thus, land-based C sequestration may become a profitable alternative to offset emissions from pastoral and agricultural areas as the price of C increases and there is greater participation in the C trading market (Cowie et al., 2012). Methods to maintain long-term SOC may vary in different pastoral or agricultural settings (e.g., afforestation and reforestation, management of crop residues, grazing land management, forest management, and revegetation; Harper et al., 2007). However, the value of woody encroachment as a revegetation technique and for enhancing C is more viable as it is less likely to be affected by drought than changes in the more ephemeral vegetative that might be altered by grazing.

In contrast, pastoralists would earn about \$A 66–470 ha^{-1} over a period of 20 years from livestock grazing alone. This assumes a carrying capacity of between 0.07 sheep ha^{-1} (in dry seasons) and 0.50 sheep ha^{-1} (in wet seasons; Robson, 1995) and gross margins of \$A 47 sheep^{-1} year^{-1} (in 2010, Khairo and Hacker, 2011). In addition to sheep grazing, pastoralists could earn an additional benefit, as much as \$A 38 ha^{-1} over the same period of time, from running cattle (assuming an average carrying capacity of 0.004 cows ha^{-1}), whose gross margin is higher than that of sheep (\$A 454 cow^{-1} year^{-1} in 2010; Khairo and Hacker, 2011). However, the likelihood of experiencing wet seasons is likely to decline given current projections of increasing numbers of low-rainfall years (Appendix B) and an increase in the frequency of droughts (Dai, 2011). Thus, the pastoral value of any increases in herbaceous biomass following ploughing is also likely to decline.

5. Conclusions

Our data suggest that the profits derived from C farming (\$A 27.40 ha^{-1} year^{-1}) are comparable to those from grazing (\$A 23.50 ha^{-1} year^{-1}; assuming a run of wet seasons). However, our study is based on the first two decades after ploughing when the greatest increase in aboveground C is expected due to re-sprouting and germinating shrubs. Responses in the following decades are likely to be substantially less, as increases in above- and belowground C stabilise. Similarly, the extent to which C farming is more profitable than grazing will depend on many things, including the relative distribution of wet and dry periods. For example, while the best possible scenario is two decades of above-average rainfall (\$A 23.50 ha^{-1} year^{-1}), a more typical scenario involving an even distribution of wet and dry periods would result in profits from grazing of only \$A 13.40 ha^{-1} year^{-1}.

Notwithstanding the apparent favourability of C farming over grazing, many uncertainties are likely to influence profitability, and therefore the uptake of C farming by the pastoral community. These include a changing price on C, the costs associated with destocking (e.g., fencing or control of feral animals), obligations for long-term C stock maintenance (Parliament of the Commonwealth of Australia, 2011) and development of more precise methods for assessing C, particularly belowground C, which take into account its spatial and temporal variability.

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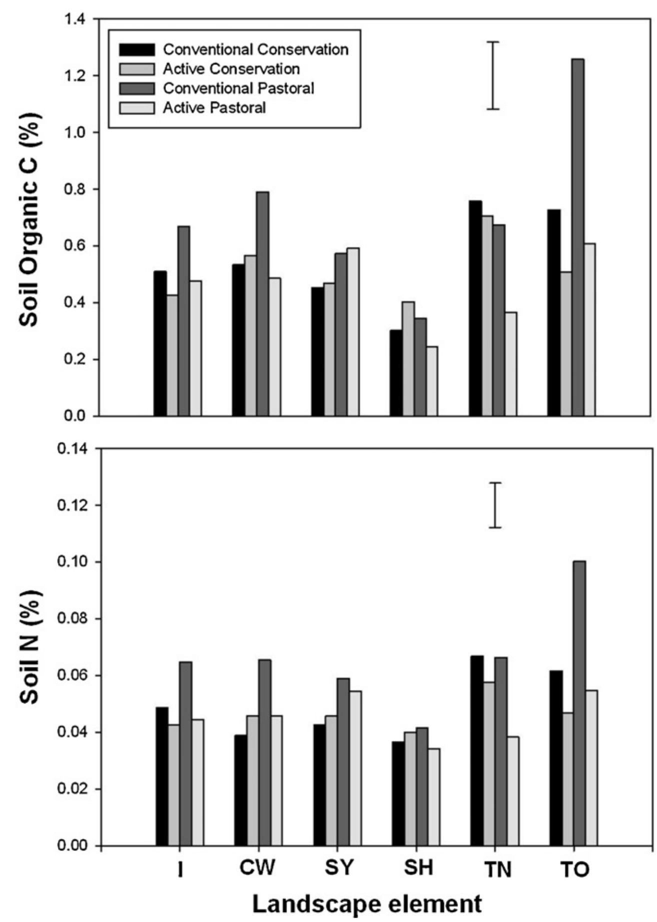
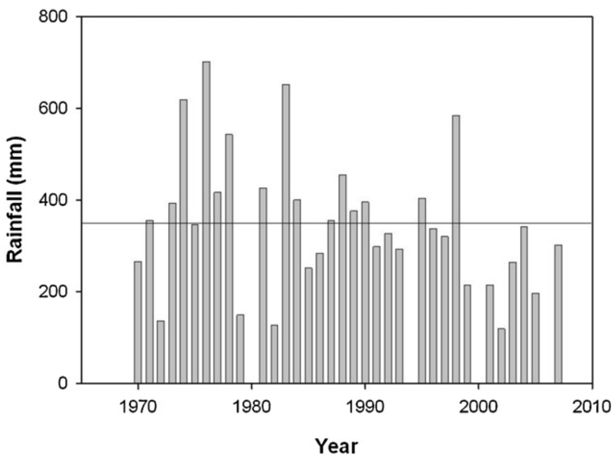
Appendix A.



Response of %SOC and soil %N to land management treatments in the different landscape elements. Error bars indicate the 5% LSD for the landscape element by management treatment interaction. Data are the integrated by depth and represent the mean 0–30 cm values. I = interspace, CW = coarse woody debris mound, SY = inverted-cone shrub, SH = hemispherical shrub, TN = N-fixing tree, TO = Non N-fixing tree. Note different scale on *y*-axis.

Appendix B.



Annual Rainfall at Fords Bridge (29°75′S, 145°43′E), the closest weather station (approximately 50 km) to Wapweelah (29°16′S, 145°26′E), over the period of 1970–2010. Solid line indicates mean annual rainfall.

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