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Carbon sequestration and biodiversity restoration potential of semi-arid mulga lands of Australia interpreted from long-term grazing exclosures

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ABSTRACT

Limited data regarding soil carbon (C) sequestration potential and biosequestration potential in arid and semi-arid environments is an impediment to appropriate policy formulation directed at greenhouse gas abatement. This paper assesses the terrestrial C biosequestration and biodiversity restoration potential of the semi-arid mulga lands of eastern Australia by measuring above and below ground C, and by making floristic biodiversity assessments in old grazing exclosures.

Grazing exclusion increased water infiltration rates and water retention capacity in the soil. Exclosures also had increased herbaceous cover and decreased bare ground. Biodiversity benefits included higher species richness and increased abundance of native grasses, many of which have become locally rare under increased grazing pressure.

The study indicates that in the absence of grazing, soil and above ground biomass, when combined, has potential carbon sequestration rates of between 0.92 and 1.1 t CO_2-e ha^{-1} $year^{-1}$ over a period of approximately 40 years. The contribution to these figures from soil C sequestration is approximately 0.18 t CO_2-e ha^{-1} $year^{-1}$, with above ground biomass contributing an additional 0.73–0.91 t CO_2-e ha^{-1} $year^{-1}$. If 50% of eastern Australia's mulga lands (half of 25.4 million ha) were managed for C sequestration and biodiversity through the control of all herbivores, then annual sequestration rates could reach between 11.6 and 14 Mt CO_2-e $year^{-1}$ which is between 2 and 2.5% of Australia's annual emissions. The potential to sequester carbon and improve biodiversity outcomes in extensive semi arid grazing lands will require significant policy shifts to encourage and reward necessary land use change.

1. Introduction

The search for potential terrestrial carbon (C) biosequestration options has focused primarily on forested landscapes or those landscapes with the potential for reafforestation. These lands tend to be located where higher and more reliable rainfall occurs. However, the potential of extensive areas of semi-arid and arid rangelands to sequester C has been receiving increasing attention because of the very large global extent of such environments (Glenn et al., 1993; Conant et al., 2001; Howden et al., 2001; Moore et al., 2001; Burrows et al., 2002; Dener et al., 2006; Harper et al., 2007; Wentworth Group of Concerned Scientists 2009). Although the current C content of semi-arid soils is low (usually around 1% or less by mass) relative to higher rainfall areas, the biosequestration potential of small increases over many millions of hectares is significant. For example, potential C sequestration in semi-arid soils of the United States has been estimated to be between 0.37 and 1.01 t CO_2-e ha^{-1} $year^{-1}$(Schuman et al., 2002) with up to 185 Mt CO_2-e $year^{-1}$ for the whole US (Lal, 2004), achieved through the use of grazing management and active restorative practices. For Australia, estimates range between 20 and 250 Mt CO_2-e $year^{-1}$ for arid and semi-arid Australia based on an assumption of restoration (Garnaut, 2008; CSIRO, 2009). Although considerable research into the interactions between grazing management and soil carbon has been undertaken in countries such as the United States (e.g. Frank et al., 1995; Schuman et al., 1999, 2002), many countries such as Australia do not have sufficient data to inform policy in this area.

The surge of interest in potential terrestrial C sequestration has to some extent overshadowed the pre-existing environmental and economic issues of productivity loss, degradation and biodiversity loss from grazed semi-arid and arid rangelands. While it is possible, and apparently intuitive, that managing these lands for increased soil C will assist in the amelioration of degradation, the linkages between environmental outcomes and C sequestration have not

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been thoroughly investigated (Beeton, 2005; Bagchi and Ritchie, 2010). An understanding of potential trade-offs between, or mutual benefits to, C sequestration and biodiversity will be essential in any policy environment (Beeton et al., 2006, p. 75; Bennett et al., 2010).

Because many arid and semi-arid environments show signs of degradation or over-utilization (Reynolds et al., 2007), restoration is desirable, but is usually too costly relative to the economic value of the land (Patrick et al., 2009). Although there has been some estimation, modelling and speculation as to the possible capacity of these lands in Australia to sequester C (see Harper et al., 2007; Garnaut, 2008; CSIRO, 2009; Fensham and Guymer, 2009), little empirical data are available for Australia and the land management required to achieve positive C accumulation and biodiversity outcomes of any changes are not known.

It may be that desirable states for ecosystem and biodiversity restoration may not lead to increased soil C or alternatively that a high soil C status can only be achieved with a less desirable ecosystem state. For example, very dense regrowth of mulga (*Acacia aneura* F. Muell) and other woody species, which is common in many semi-arid woodlands and shrubland of eastern Australia, may result in relatively high above ground biomass and moderate to high soil C content but little ecosystem process or habitat value for rare and threatened species. This situation could also change the hydrological balance either positively (greater soil moisture availability) or negatively (reduced run off for downstream needs). Alternatively, a lower density of woody vegetation and high herbaceous biomass coverage may be desirable for biodiversity values, but with unknown C storage potential. Further layers of complexity are added when issues of total grazing pressure and fire management, and in the case of the mulga lands specifically, the practice of mulga fodder harvesting (Page et al., 2008) are taken into account (Beeton et al., 2006, p. 40).

An opportunity exists to determine relationships between C biosequestration and some attributes of biodiversity because of the presence of long-term grazing exclusion experiments established across the mulga lands of south western Queensland since the 1960s. The mulga lands are representative of many of the environmental issues facing rangelands in general, and they have been subject to a reasonable level of formal research over many decades (Page et al., 2008). The grazing exclusion research since the 1960s replicates the effects of reduced or negligible grazing, with individual sites established for periods ranging from 13 to more than 40 years. This study does not aim to determine, nor does it assume what the pre-European vegetation state would have been, due to the variable and modified nature of ecosystems in the mulga lands (see Witt and Beeton, 1995; Witt et al., 2006, 2009). The outcomes from, and implications of, these exclusion studies have remained poorly reported with very few peer reviewed publications available. The result has tended to be that many of these exclosures have remained unmonitored, and at risk of deterioration and loss from the collective research consciousness.

The purpose of this study is to harness the ecological information from a sample of long-term grazing exclusion experiments to explore the impact of reduced grazing pressure on woody biomass, floristic biodiversity and soil C as well as to identify any synergies between them.

2. Methods

2.1. Study area

This study was conducted in the mulga (*Acacia aneura*) lands bioregion (Sattler and Williams, 1999, Fig. 1) in south west Queensland, Australia, which covers approximately three quarters of the mulga lands of eastern Australia. Although no sites were investigated in New South Wales, the ecosystems in Queensland are also largely representative of those occurring in north-west New South Wales. The region has a semi-arid climate with average annual rainfall ranging from almost 500 mm in the east to 150 mm in the west. Rainfall is highly variable with extended periods of below average rainfall punctuated by occasional short periods of above average rainfall. Temperatures are very warm to hot in the summer months (average maximum January temperatures above 36 °C) with cool winters (average minimum temperatures in July less than 6 °C), frosts are rare in the region.

The soils of the region range from red and brown loams (Kandosols) and texture contrast soils (Chromosols) to siliceous sands (Tenosols) in the uplands, with grey and brown cracking clays (Vertosols) dominating the alluvial areas; details of soils and geomorphology are available from Dawson and Ahern (1974) and Ahern and Mills (1990). The vegetation is dominated by woody plants (trees and shrubs) and plant association ranges are strongly linked to the geology and soil characteristics. Much of the vegetation is open to tall shrublands, open woodlands and woodlands that are generally dominated by mulga, with *Eucalyptus* (particularly *E. populnea* and *E. melanophloia*) becoming more dominant in the east. Some *Eucalyptus* species dominate the vegetation in places such as riparian systems, flood plains or other run on areas (Neldner, 1984). Most of the region is not suited to crops and the bulk of the land is used for sheep and cattle grazing with individual enterprises ranging typically between 35,000 and 65,000 ha in the central and eastern areas and over 100,000 ha in the west.

Grazing in the area was initially restricted to drainage lines and was subsequently massively extended by the tapping of artesian water (Gasteen, 1986), which permitted increased domestic grazing pressure as well as numbers and grazing pressure of kangaroos. The consequence was unquantified, but undoubtedly significant loss of above and below ground biomass and C. Rabbits reached the area in the 1890s and up to the 1950s had a significant impact on vegetation.

2.2. Field site descriptions

Nine grazing exclosures were sampled across the region (Fig. 1). The exclosures were erected between 1966 and 1996, with average size of approximately 2000 m^2 and were originally used for a range of experiments involving stock removal and thinning of woody vegetation (see Table 1 for overview). Monitoring in all of these exclosures had ceased prior to our study with the exception of one site (Wallen 5). All exclude domestic stock with some designed to exclude all native and feral grazing animals. Sites were sampled during two field trips on the 8–27 May and 10–30 June 2009. This had implications for some results as rain fell across parts of the region between the two sample periods.

Information on exclosures was retrieved from the Queensland Herbarium, communication with ex-managers and various researchers. Time and resource constraints limited the number of exclosures used in this study to 9 sites with selection based on the criteria of wide spatial and temporal (i.e. time since construction of the exclosure) distribution of the exclosures and that they were located in mulga dominated land systems.

The Currawinya and Mariala sites are located on what are now national parks that have had domesticated cattle and sheep removed. Both sites were previously grazing enterprises since approximately the 1860s. Mariala was gazetted as a national park before 1982 and Currawinaya in 1992. However, feral (predominantly goats) and native (red and grey kangaroos) herbivores are still present to such an extent that they exert considerable grazing pressure (Page, 1997). All other sites have been subject to continuous grazing.

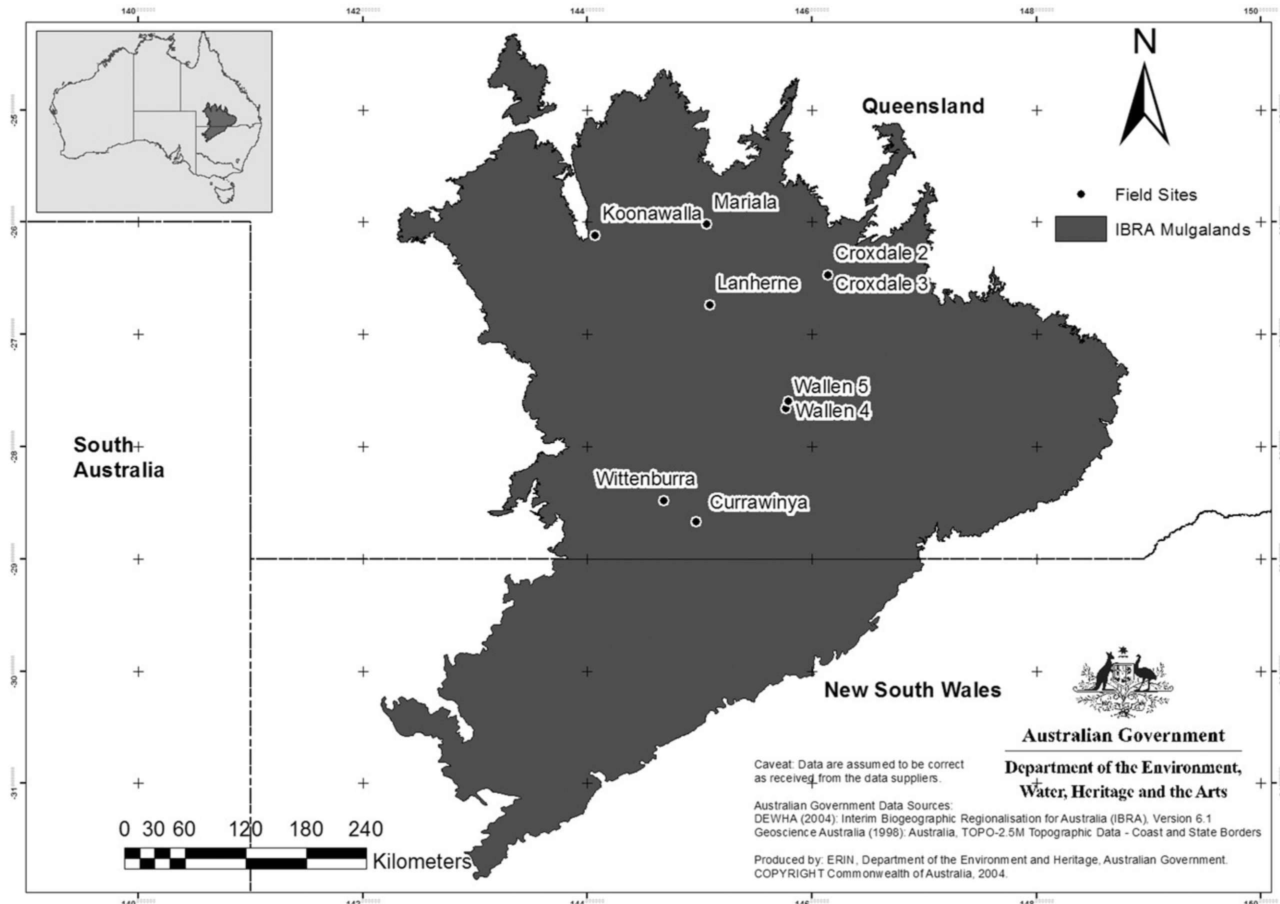

Fig. 1. Location of the mulga lands biogeographic region in south western Queensland, major towns, and field sites.

2.3. *Sampling design, field procedures and analysis*

The primary objective of the study was to ascertain how the removal of grazing pressure affects C sequestration (above and below ground) and biodiversity within the exclosures. Thus, we used a paired design approach which compared data from inside and outside each exclosure. We have assumed that the paired sites were the same at the time when fences were erected, and that any differences in C, both above and below ground, at the time of sampling will represent either a gain or loss which results from the reduction in grazing. The rates of sequestration or loss determined are the average of the entire period, and, using this approach, we are unable to predict what new equilibrium level will ultimately be achieved.

Because exclosures vary in size, sampling was standardized to an area of 10 m × 40 m. Three parallel 40 m transects, each five metres apart, were used for systematic sampling (see Fig. 2). The sampling area located outside the exclosure was established at a minimum distance of 20 m from the exclosure and was selected to minimise the differences in environmental variables (slope, soil type, etc.).

2.3.1. *Soil sampling*

Ten soil core replicates were taken from inside and a further ten from outside all sites (20 per site in total). Soil cores were taken

Table 1
Site information ordered by average annual rainfall.

Site name	Site no.	Date fenced and type[a]	Size (m)	Lat. (°S)	Long. (°E)	Mean rainfall[b] (mm)
Wittenburra	10	1966 (stock only)	20 × 50	28.48	144.68	290
Currawinya	1	1993 (full)	40 × 20	28.67	144.97	292
Koonawalla	9	1966 (stock only)	50 × 20	26.12	144.07	310
Wallen 5	5	1996 (full)	30 × 100	27.66	145.77	368
Wallen 4	4	1980 (full)	50 × 50	27.60	145.79	374
Lanherne	7	1984 (full)	50 × 50	26.74	145.09	378
Mariala	8	1982 (full)	50 × 50	26.02	145.06	424
Croxdale 2	2	1981 (full)	50 × 50	26.47	146.14	469
Croxdale 3	3	1981 (full)	50 × 50	26.47	146.14	469

[a] With the exception of Currawinya information compiled by personal communication Ms Jenny Silcock (Queensland Department of Environment and Resource Management), Dr. Peter Johnston (Queensland Department of Primary Industries and Fisheries) and Dr. Ken Hodgkinson (Retired).
[b] Based on interpolated data as not all these locations have a nearby rainfall observation point.

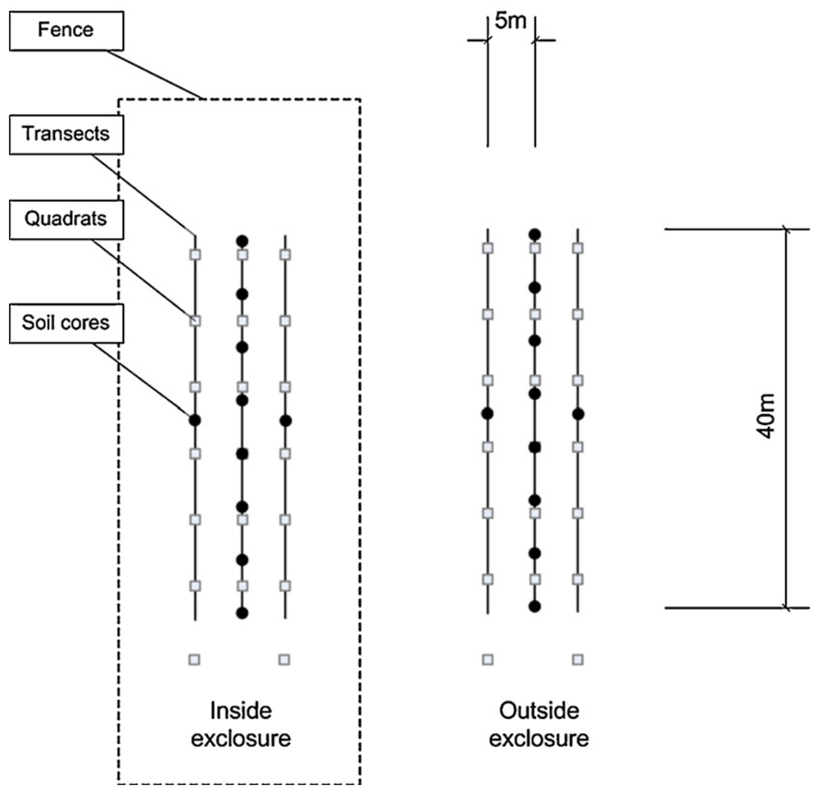

Fig. 2. Representation of the sample design and configuration of transects inside and outside grazing exclosures. There were a total of 10 soil cores and 20 quadrats inside the exclosures and this was repeated outside.

to a depth of 30 cm with a diameter of 5 cm (Eamus et al., 2000). Cores were collected at 5 m intervals (eight cores from the middle transect and two cores from the centre of each side transect). In the laboratory wet soil weights were recorded then samples were left to dry at 40 °C in an oven for 4 days. Dry weight was recorded then soil was sieved to 2 mm and weight of rocks, leaf litter and roots were noted for each core. Bulk density of soil was calculated by excluding the weight of rock from the dry weight of soil and subtracting the volume of rock from the volume of the core using a constant density for rock fragments of 2.6 g cm^{-3}. In some instances in the rocky soils, displacement of rocks outward by the corer, or compaction of the rock and soil in front of the core resulted in incorrect low values. All bulk density results below 1.0 g cm^{-3} were discarded.

A subsample of sieved soil was crushed, thoroughly mixed, and used for elemental and stable isotopic analyses (both C and nitrogen (N)). Carbon (δ^{13}C) and nitrogen (δ^{15}N) values are reported as parts per thousand (‰) deviations from the isotopic composition of the international V-PDB (carbon) and AIR (nitrogen) standards with an uncertainty of ±0.1‰. Soil C, when reported in terms of t ha^{-1} (either as C or as CO_2 equivalents (CO_2-e)), are to a depth of 30 cm. Soil moisture was determined by using the percent weight of water in soil, excluding the weight of rocks.

2.3.2. Above ground woody vegetation

The areas sampled for this study are relatively small (i.e. 400 m^2) and for this reason most results are discussed in the context of variables that could be confidently measured at the scale of sampling.

All trees inside the 400 m^2 sample area were identified and measured to estimate above ground woody biomass. Large and small trees were dealt with differently. Diameter (at 30 cm above ground level) was only recorded for individuals when it exceeded 4 cm. Plant height determined by clinometers, and canopy measurements were also obtained for these trees (Burrows, 1974; Snowdon et al., 2002). For smaller trees (diameter less than 4 cm at 30 cm above the ground), only species and height were recorded. To determine standing biomass for trees, predictive allometric equations for mass were used. For large trees, methods described by Burrows et al. (2000, 2002) were followed, while for smaller trees (diameter <4 cm) methods described by Harrington (1979) were used to convert tree height to biomass. These relationships are based on regressions from sampled mulga and 'general eucalypts', as described in Eamus et al. (2000). The woody mass was then multiplied by a standard factor of 0.5 to convert dry wood weight to C weight (Ragland et al., 1991), and above ground woody biomass was then extrapolated to t ha^{-1} where appropriate.

2.3.3. Saturated hydraulic conductivity

Infiltration rates are indicative of soil functionality at scales relevant to the size of the exclosures (Stavi et al., 2009). Water infiltration rates are an important physical attribute of the mulga bioregion, as water is a major limiting factor for primary productivity and C biosequestration. The site was stratified into ground cover type (bare or vegetated) and duplicate infiltration rate measurements were taken in each stratum at set time intervals (0, 1, 2, 5, 10, 15, 30 and 60 min). A single ring infiltrometer (40 cm ⌀ × 20 cm high) was placed 2 cm into the soil with little or no disturbance to the surface. Both sides of the ring were then sealed with loamy soil to prevent water escaping from the sides of the infiltrometer. Water (7.5 l) was slowly poured into the ring onto a piece of canvas to reduce soil surface disturbance. The canvas was removed and the first measurement was taken when the water settled (after approximately 30 s). Field-saturated hydraulic conductivity was calculated using the method of Reynolds and Elrick (1990).

2.3.4. Herbaceous, litter and biological soil crust cover

Herbaceous plant richness (the number of species) and abundance (the cover of each species) were recorded in 20 m × 1 m

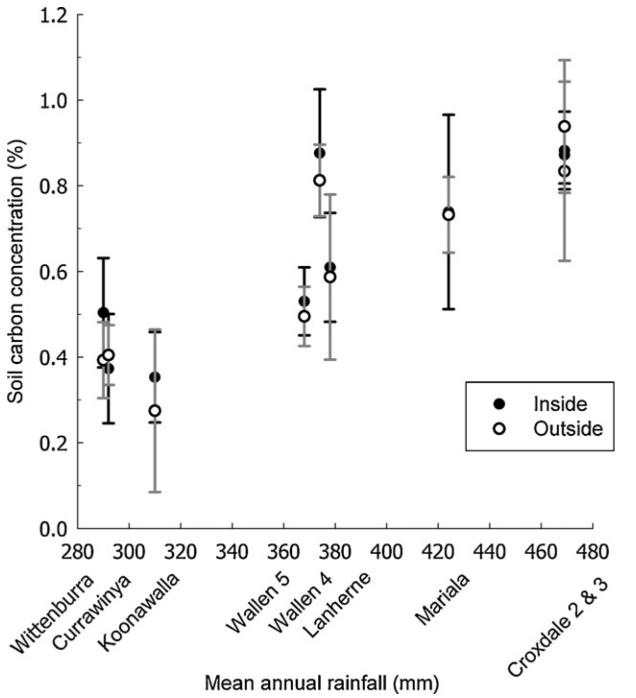

Fig. 3. Average ($\pm 1\sigma$ error) soil carbon concentration inside and outside exclosures for all sites plotted against mean annual rainfall.

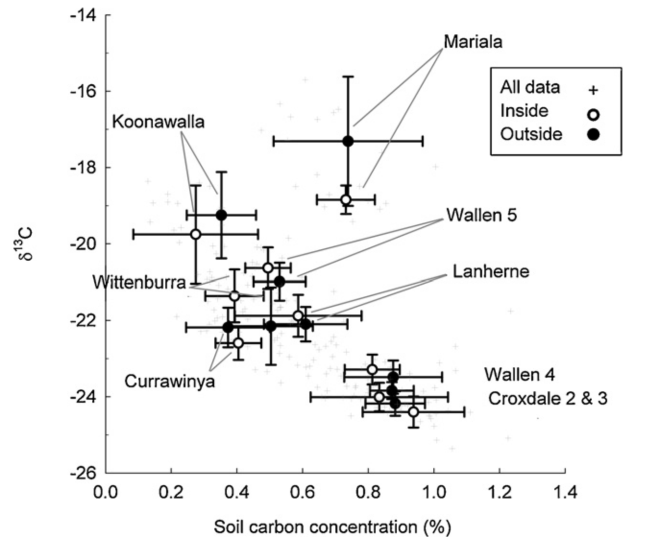

Fig. 4. Relationship between carbon concentration and δ^{13}C values for soil carbon for all samples, with the average for inside and outside also plotted with 1σ errors. Site names are indicated.

quadrats at 5 m intervals inside and outside each exclosure. Litter cover, bare ground, small rocks (<5 mm ⌀) and biological soil crust cover were estimated for each quadrat (Tongway and Hindley, 2004; Williams et al., 2008).

2.3.5. Data analysis

For most variables measured in this study (soil C, soil moisture, bulk density, herbaceous cover, etc.) we used paired *t*-tests to compare the mean values collected from inside and outside the exclosure. Student's *t*-test was used to compare conditions within the exclosure to outside at a single site. A non-parametric test (Mann–Whitney *U*) was used to compare the means of C and N isotope composition.

In the analysis of C, one individual replicate value (of the 10 replicates measured) from each of three sites (Currawinya-outside, Mariala-outside, Croxdale-outside) had unexpectedly high values. These replicate points were identified as outliers using a discordancy test for a single upper outlier (Moran and McMillan, 1973), and omitted from the analysis. The three points remain in the descriptive statistics and are included in the graphs of data.

3. Results

3.1. Soil carbon and nitrogen

For all sites and every individual core sample, C contents ranged from 0.13% to 1.32%, while N contents ranged from 0.018% to 0.081% and C:N ratios from 6.1 to 20.6. There is a clear relationship between mean annual rainfall and the average C contents of soils both inside and outside exclosures, with mean C content increasing quasi-linearly as rainfall increases from 0.25–0.5% at <350 mm to 0.78–0.95% at >450 mm (Fig. 3). Wallen 4 soils contain more C than site average annual rainfall alone would predict, reflecting the higher clay content of soils at this site relative to other sites in the study (except the Mariala National Park site).

Across all sites, average soil C concentration within the exclosures (0.64%) was not significantly higher than the soil outside (0.61%) ($p = 0.12$) (Tables 2 and 3). However, if Mariala and Currawinya are excluded (the latter two have been released from grazing pressure by stock over at least the last 15 years) the soil C contents inside the exclosures (0.66% C) are significantly higher ($p = 0.039$) than outside (0.62% C). Average increases of 28–29% in the percentage of soil C are evident at two of the driest sites (Wittenburra and Koonawalla), where (Table 2). These exclosures have been in existence for approximately 44 years. The total soil C stocks at these sites outside the exclosures are low (Wittenburra 15, Koonawalla 11 t ha^{-1}), but have increased inside the exclosures by 4 t ha^{-1} at Wittenburra and 8 t ha^{-1} at Koonawalla (15 and 29 t CO_2-e ha^{-1}) (Table 2). This equates to an increase in average C accumulation rate of 0.13 t C ha^{-1} year^{-1} or 0.50 t CO_2-e ha^{-1} year^{-1} over the lives of the exclosures.

For the sites where treatments have been maintained (i.e. excluding the national parks, Mariala and Currawinya) the C accumulated inside the exclosures amounts to slightly more than 1.5 t ha^{-1} of C after an average of 30 years (an annual rate of accumulation in the soil $\leq$0.05 t ha^{-1} year^{-1} or 0.18 t CO_2-e ha^{-1} year^{-1}). In the case of Currawinya ($\sim$15 years of exclusion) and Croxdale 3 ($\sim$28 years of exclusion), small decreases in C content have occurred (Table 2), though these are not statistically significant.

Further information on the dynamics of the C and N cycles at these sites is provided by the C and N isotope composition as well as C:N ratio. Soil C isotope compositions (δ^{13}C values) in the mulga lands, where grasses dominantly use the C_4 photosynthetic pathway, are affected by the relative proportions of grass (C_4, which average isotopic values of around −12‰) versus tree (C_3 averaging approximately −26‰) derived C inputs. There is a clear relationship between higher soil C contents, associated with wetter sites, and a higher proportion of woody biomass (Croxdale 2 and 3, and Wallen 4) contributing to the soil δ^{13}C values (Fig. 4). Higher (less negative) soil δ^{13}C values are associated with increasing aridity and can be attributed to less woody vegetation contributing to soil C, but also to the observation that C_3 plants tend to exhibit slightly higher δ^{13}C compositions in more arid environments. Comparison of δ^{13}C values inside and outside the exclosures does not indicate a significant shift in δ^{13}C towards higher values (Table 4) (the exception being at Mariala, discussed below), despite the proportion of grass cover having increased inside the exclosures compared to outside. This is consistent with the relatively modest gains in soil C inside

Table 2
Average soil bulk density, carbon content, and saturated hydraulic conductivity (K_{sat}).

Site name	BD g cm^3		Soil C% (t CO_2-e ha^{-1})		K_{sat} cm s^{-1} × 10^{-3} bare (veg)	
	Inside	Outside	Inside	Outside	Inside	Outside
Wittenburra	1.17	1.24	0.50 (18)	0.39 (15)	0.31 (8.3)	0.83 (0.66)
Currawinya	1.25	1.24	0.37 (14)	0.40 (15)	0.83 (2.3)	0.54 (1.4)
Koonawalla	1.22	1.35	0.35 (13)	0.27 (11)	0.30 (1.3)	0.20 (0.30)
Wallen 5	1.25	1.22	0.53 (20)	0.49 (18)	0.42 (1.9)	0.38 (0.74)
Wallen 4	1.07	1.10	0.88 (28)	0.81 (27)	1.42 (6.1)	0.71 (2.9)
Lanherne	1.20	1.21	0.60 (22)	0.59 (22)	0.22 (0.81)	0.28 (0.81)
Mariala	1.16	1.12	0.74 (26)	0.73 (25)	0.97 (3.2)	0.97 (4.4)
Croxdale 2	1.13	1.19	0.87 (29)	0.83 (30)	0.19 (6.7)	0.36 (4.7)
Croxdale 3	1.05	1.11	0.88 (28)	0.94 (31)	7.08 (6.9)	0.86 (6.7)
Average all sites	1.17	1.20	0.64 (22)	0.61 (22)	1.3 (4.2)	0.57 (2.5)
Average old sites[a]	1.20	1.30	0.43 (16)	0.33 (13)	0.31 (4.8)	0.52 (0.48)
Average young sites[b]	1.16	1.17	0.70 (24)	0.68 (24)	1.6 (4.0)	0.59 (3.1)

[a] Wittenburra and Koonawalla (44 years).
[b] Currawinya, Wallen (4 and 5), Lanherne, Mariala, Croxdale (2 and 3) (average 23 years).

Table 3
Summary statistics and paired *t*-test for the measured variables.

Environmental variables	Unit	Inside mean (SD)	Outside mean (SD)	*N*	*t*	*P* value[b]
Soil carbon concentration	%	0.64 (0.21)	0.61 (0.23)	9	1.7	0.13
Soil carbon content	t ha^{-1}	23 (6.2)	22 (6.9)	9	1.7	0.49
Carbon (live trees <15 cm in ⌀)	t ha^{-1}	5.3 (4.6)	1.1 (2.2)	9	2.4	0.044*
Bulk density	g cm^3	1.17 (0.07)	1.20 (0.08)	9	−1.7	0.13
Hydraulic conductivity[a]	cm s^{-1} × 10^{-3}	2.73 (2.86)	1.54 (1.87)	18	2.19	0.043*
Soil moisture (sites pre rain)	% w/w	1.65 (0.73)	1.91 (0.96)	5	−2.2	0.097
Soil moisture (sites post rain)	% w/w	12.5 (2.2)	7.23 (4.1)	4	3.4	0.044*
Leaf litter in soil	% w/w	0.18 (0.12)	0.11 (0.10)	9	1.5	0.166
Root mass in soil	% w/w	0.04 (0.05)	0.02 (0.02)	9	1.79	0.11
Litter ground cover	%	29.1 (19.4)	26.6 (22.8)	9	0.37	0.72
Bare ground cover	%	34.8 (16.6)	54.6 (21.0)	9	−2.4	0.043*
Biological soil crust cover	%	31.4 (8.9)	30.1 (21.2)	9	0.19	0.85
Projected herbaceous cover	%	40.6 (13.3)	21.7 (9.1)	9	3.77	0.006**
Plant species richness	species/site	10.1 (5.7)	8.1 (3.6)	9	1.3	0.22
Total number of trees	trees/400 m^2	45 (58)	8.7 (8.1)	9	2.1	0.065

[a] Data for bare and vegetated sites.
[b] $^*P < 0.05$, $^{**}P < 0.01$.

exclosures at most sites, and suggests that most additional C_4–C is likely to have decomposed above the soil surface, and at a faster rate than C_3-derived woody C. Water stress may have been reduced in the exclosures, however the extent to which this is influencing the soil carbon isotope values cannot be determined from these data due to the much larger influence of C_4 inputs which are likely to have increased in the exclosures.

The Mariala data plot above the trend identified for other locations (Fig. 4) suggesting a greater proportion of C_4-derived C at this site, and therefore a greater proportion of grass biomass relative to tree biomass. While there has been no significant increase in soil C inside the exclosure at Mariala, there has been a significant increase in the δ^{13}C value of the soil C. This suggests that there has been an increase in the proportion of C_4 (grass) derived C entering the soil as a result of exclosure establishment, but that any potential gains in soil C stocks have been offset by the increased decomposability of the C in the soil. If it is assumed that pure C_4 derived soil C isotope signatures for this semi-arid system is −12‰ and that for C_3 −24.5‰, then the Mariala result indicates that inside the exclosure almost 60% of soil C has been derived from C_4 grasses.

Nitrogen content ranges from 0.018 to 0.081% (mean (0.045% ± 0.012 1σ) and δ^{15}N value from +2.9 to +13.0‰ (mean +8.1‰ ± 1.84 1σ) (Table 4, Fig. 5). Significantly, the average δ^{15}N values at sites Koonawalla and Wittenburra, which recorded the largest increase in C stocks, are the lowest of any sites. Low δ^{15}N values are suggestive of *Acacia* N fixation or potentially higher inputs from biological soil crusts. It may be that greater available N at the Koonawalla and Wittenburra sites is the reason why significant C

Table 4
Average carbon and nitrogen isotopic values and C:N ratios inside and outside exclosures.

Site	δ^{13}C (‰)		δ^{15}N (‰)		C/N	
	Inside	Outside	Inside	Outside	Inside	Outside
Wittenburra	−22.1	−21.4	6.6	5.9	10.4	9.2
Currawinya	−22.2	−22.6	8.5	7.8	14.2	13.9
Koonawalla	−19.3	−19.8	6.1	6.1	10.8	8.8
Wallen 5	−21.0	−20.6	8.5	7.1	12.2	12.4
Wallen 4	−23.5	−23.3	9.9	9.7	15.2	14.3
Lanherne	−22.1	−21.9	8.9	7.9	12.9	13.1
Mariala	−17.3	−18.8	8.2	7.9	14.6	13.7
Croxdale 2	−23.8	−24.0	9.0	8.9	17.2	15.7
Croxdale 3	−24.2	−24.4	9.4	9.3	17.1	16.8
Average	−21.7	−21.9	8.3	7.8	13.8	13.1

Table 5
Total number of living and dead standing trees in 400 m^2 inside and outside of the exclosures.

Site name	Small trees >15 cm Ø[a]				Large trees <15 cm Ø			
	Inside		Outside		Inside		Outside	
	Total	(Dead)	Total	(Dead)	Total	(Dead)	Total	(Dead)
Wittenburra	4	(2)	1		4	(4)		
Currawinya	6		4		6		1	
Koonawalla	7		1				2	
Wallen 5	28		16					
Wallen 4	59	(37)	0		1	(1)	5	(4)
Lanherne	41		0				2	
Mariala	2		0		3		5	
Croxdale 2	182		18		5	(5)	5	
Croxdale 3	50		8		11	(2)	10	

[a] This is the diameter of the trunk/stem at 30 cm above the ground.

accumulation inside the exclosures was recorded at these sites and not at other sites. These sites also had the lowest C:N ratios and recorded the greatest increase in C:N ratio in the exclosures relative to outside, suggestive of increased N use efficiency at these sites (Table 4).

3.2. Carbon in above ground woody biomass

The removal of grazing pressure resulted in clearly visible regeneration of trees (primarily mulga) across many sites. Most sites contained a mixture of living and dead trees, and most of the standing dead trees appear to have succumbed in the extended dry period that affected much of eastern Australia between 2001 and 2007 (Table 5), highlighting that tree death tends to occur episodically rather than continuously and this has implications for determining standing biomass (discussed below).

Significant regeneration occurred inside the exclosures, particularly at Wallen 4, Lanherne and both of the Croxdale sites (Table 5). Wittenburra and Koonawalla are potentially indicating that, at the driest end of the mulga lands, opportunities for tree recruitment are less frequent, as there has been little successful recruitment after more than 40 years.

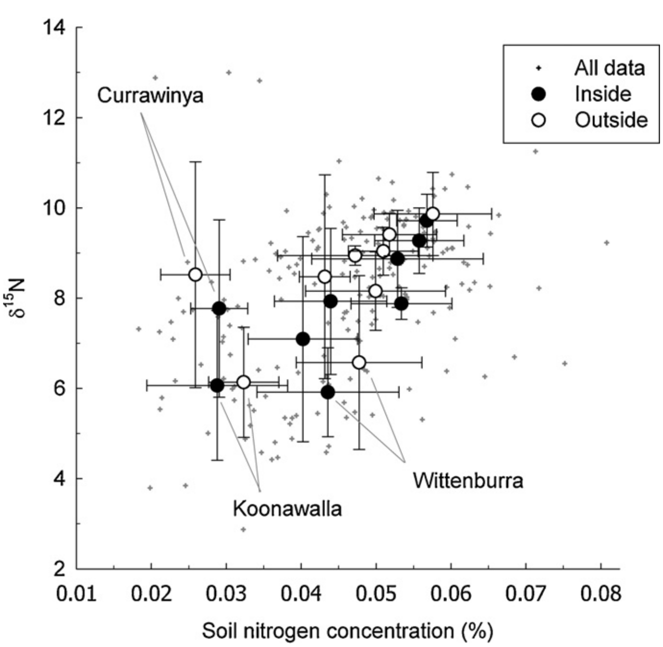

Fig. 5. Relationship between nitrogen concentration and $\delta^{15}N$ values for soil nitrogen for all samples, with the average for inside and outside also plotted with 1σ errors.

The limited regeneration of trees at Mariala is most likely a reflection of the outcome of competition for resources between the herbaceous layer (which is very dense at this site) and germinating tree species. Croxdale 2 recorded close to a 10-fold increase in regeneration inside versus outside the exclosure, indicating (in contrast to Mariala) the very strong potential for grazing pressure to alter the competition between tree and herbaceous layers.

Woody biomass conversions to sequestered C are presented in Table 6. Two sets of data show the influence of tree mortality on calculations. As reflected in the tree count data (Table 5), recruitment has generally led to increases in C stocks inside the exclosures. The difference between estimations based on living only, or living with dead trees (Table 6), is indicative of the problems of accounting for above ground C in dynamic semi-arid systems. If only living trees are counted then the estimated increase in C (in woody vegetation above ground) is 4.2 t ha^{-1} over approximately 28 years (based on the difference between the average of all small trees inside and outside exclosures). This equates to an annual accumulation of 0.15 t C ha^{-1} $year^{-1}$ or 0.56 t CO_2-e ha^{-1} $year^{-1}$. If the national park sites of Mariala and Currawinya sites are excluded (as discussed in the section on soil C above), then the result increases substantially to 5.8 t C ha^{-1} over 30 years (0.19 t C ha^{-1} $year^{-1}$ or 0.71 t CO_2-e ha^{-1} $year^{-1}$). In the far west of the region, rates are as low as 0.05 t C ha^{-1} $year^{-1}$ at Wittenburra and 0.07 t C ha^{-1} $year^{-1}$ at Koonawalla.

Due to the long periods required for the breakdown and decay of standing dead trees in the mulga lands (particularly *Acacia* spp.). The amount of C accumulated in woody living and dead vegetation due to grazing exclusion is estimated at 4.3 t ha^{-1} over 28 years equating to an annual rate of 0.15 t C ha^{-1} $year^{-1}$, and is only slightly more than estimates based on only living trees.

Increased recruitment of mulga, frequently following periods of good rainfall does result in a dense cohort of young trees. In the absence of grazing pressure, particularly from sheep, the competitive effect between cohorts leads to thinning and potential loss of mature trees. Although there were relatively few sites able to be investigated, and there was the problem of sample area available in the exclosures (discussed previously), the data in Table 5 (tree counts) and Table 6 (C stocks) show the impact of competition on large trees in the exclosures. In the cases of Wittenburra, Wallen 4 and Croxdale 2 all large trees inside the exclosure had recently died. The large trees contain vastly more C than the large numbers of small trees. To avoid the complications associated with large tree distribution and the issue of tree mortality, estimates of C used for discussion purposes are based only on the living biomass results. These are the data for small live trees presented in Table 6. It should be noted that to some extent C gains through regeneration in the absence of grazing may be negated by mortality in larger

Table 6
Biomass in tonnes of carbon per hectare (t ha^{-1}) of live, and a combination of live and dead trees in two categories (small and large) (based on a sample area of only 400 m^2).

Site	Small trees				Large trees			
	<15 cm Ø inside[a]		<15 cm Ø outside		>15 cm Ø inside		>15 cm Ø outside	
	Live	Total	Live	Total	Live	Total	Live	Total
Wittenburra	2.1	4.1	0	0	0	9	0	0
Currawinya	3.3	3.3	6.9	6.9	17.1	17.1	5.2	5.2
Koonawalla	4.4	4.4	1.3	1.3	0	0	0	10.1
Wallen 5	3	3	0	0	0	0	0	0
Wallen 4	1.6	11.5	0	0	0	14.1	7.1	7.1
Lanhern	15.4	15.4	0	0	0	0	12.6	12.6
Mariala	1.4	1.4	0	0	17.5	17.5	24.8	24.8
Croxdale 2	7.6	7.6	0.3	0.3	0	21.3	44.3	44.3
Croxdale 3	9.2	9.2	1	1	23.6	26.4	48.1	48.1
	5.3	6.7	1.1	1.1	6.5	11.7	15.8	16.9

[a] The small tree category of <15 cm diameter at 30 cm above ground is used as an approximation to capture those tress that have established since grazing exclusion.

trees which contain significantly more C, though these will decay slowly.

3.3. Soil physical characteristics

Soil bulk density averaged across all sites was 1.17 g cm$^{(3}$ ($\pm$0.07 1σ) inside the exclosures, while the average outside the exclosures was 1.20 g cm$^{(3}$ ($\pm$0.08 1σ) (Table 3). These values were not significantly different (P = 0.58).

The capacity of semi-arid systems to absorb often infrequent and small rainfall events is a critical factor in ecosystem processes. Long-term grazing exclusion significantly (P = 0.04) increased saturated hydraulic conductivity across all sites (Table 3). Average saturated hydraulic conductivity was 2.8 $\times$ 10$^{(3}$ cm s$^{(1}$ within the exclosure, and 1.5 $\times$ 10$^{(3}$ cm s$^{(1}$ outside. Vegetated patches have the greater saturated hydraulic conductivity with means of 4.2 $\times$ 10$^{(3}$ inside and 2.5 $\times$ 10$^{(3}$ cm s$^{(1}$ outside, compared to bare patches with 1.3 $\times$ 10$^{(3}$ inside and 0.57 $\times$ 10$^{(3}$ cm s$^{(1}$ outside (Table 2). The effect of these hydraulic conductivity differences on infiltration and water storage during rainfall is further accentuated by the greater proportion of vegetated area within the exclosure. The net effect of the greater hydraulic conductivity, and greater proportion of vegetated area, is shown dramatically in the water content of soils at the four sites sampled after rain (Wittenburra, Koonawalla, Lanherne and Mariala) (Fig. 6). For these sites the average gravimetric water content of the soil inside the exclosures (14.5%) is close to double that observed outside (7.8%).

These observations affecting infiltration are associated with other soil features, specifically leaf litter and root mass in the soil, biological soil crust and herbaceous cover and the extend of bare ground all of which showed significant difference inside and outside of the exclosures (Table 3).

3.4. Vegetation

The absence of livestock has lead to many soil changes that allow for more vigorous herbaceous growth in addition to the direct reduction in removal of plant material by grazing. For all sites the herbaceous cover is approximately twice as much inside the exclosures at 40.6% ($\pm$15.1 1σ) compared to only 21.7% ($\pm$9.2 1σ) outside (Table 3, Fig. 7).

Of the 52 plants (excluding trees) identified across all sites, 33 were found both inside and outside the exclosures. However, 15 were confined to the exclosures and only two were exclusively found outside (data not presented). The general trend is for increased numbers of grass species and herbaceous cover inside the exclosures.

There are more species recorded inside the exclosures (with the exception of Currawinya and Croxdale 3). Wittenburra and Koonawalla (both in the low rainfall part of the region) had considerably more species in the exclosures compared to outside.

It could be assumed that removal of grazing pressure would permit the establishment of species that are sensitive to grazing. Some species may occur in grazed landscapes but at lower abundance than prior to grazing. The data available from these nine small grazing exclosures do not provide robust data for statistical analysis, and there only appears to be a weak relationship between the age of exclosures and the 'increase' in numbers of plant species inside the exclosures (Fig. 8). That said, with the exception of Croxdale 3, it does appear that exclusion has led to an increase in the number of plant species (15 found exclusively within exclosure, with 3 found exclusively outside).

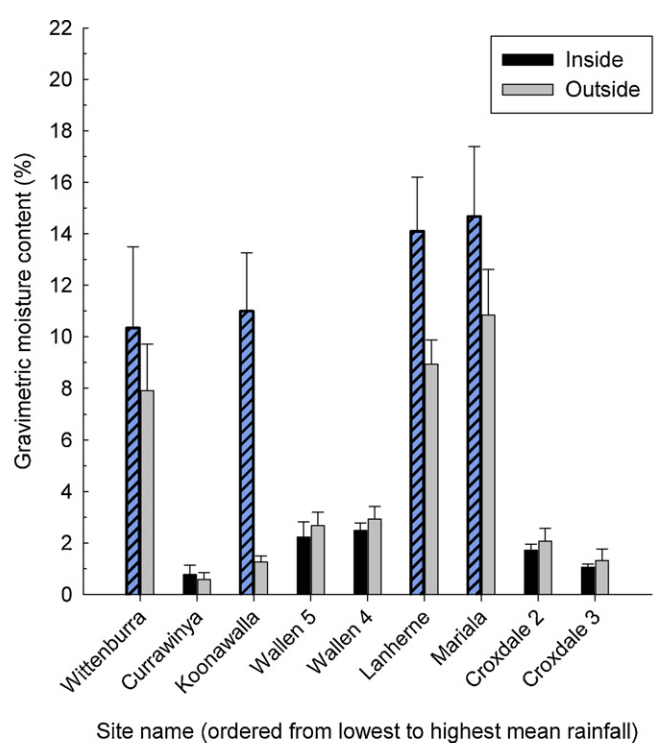

Fig. 6. Average ($\pm$1σ error) soil gravimetric moisture content (note that the hatched bars represent the sites that received rainfall within two weeks prior to sampling).

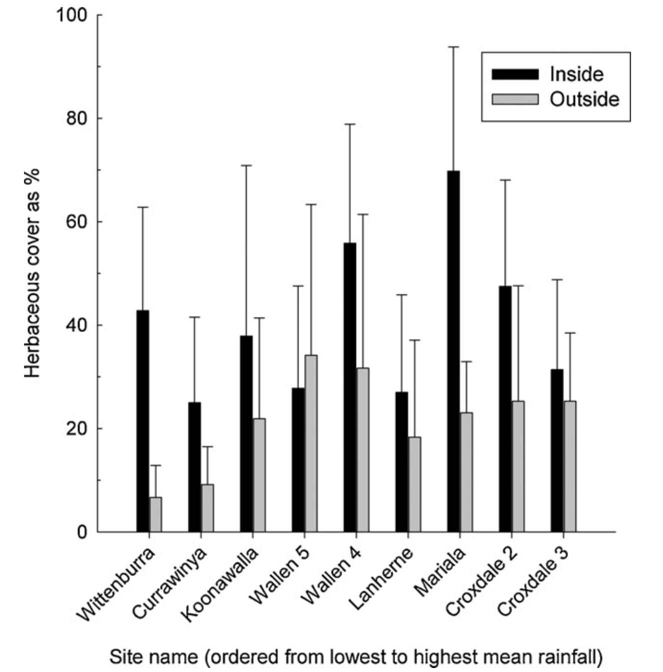

Fig. 7. Average ($\pm 1\sigma$ error) projected herbaceous cover (as a percent) inside and outside the exclosures.

4. Discussion

4.1. Grazing exclusion, soil carbon and above ground biomass

Grazing exclusion resulted in a significant increase in soil C concentration from 0.62% to 0.66% on sites where the treatments (grazed vs non-grazed) were sustained up to the time of sampling. Increases were most pronounced at the long-term sites in this study. Similar rates of C accumulation have also been reported for a 30-year-old exclosure at Tyrone (Witt, 1998). We attribute the increased soil C to increased primary biological productivity within the exclosures; the net result of the reduction in grazing pressure, and the increased water availability resulting from increased infiltration rates within the exclosure.

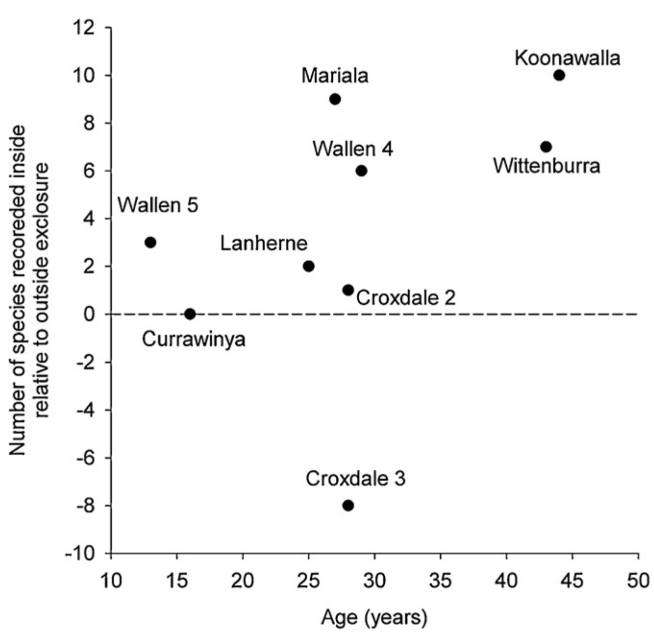

Fig. 8. The relationship between age of exclosure and the difference in herbaceous species identified inside the exclosures versus outside the exclosures.

Following grazing exclusion, soil C increased at 0.18 t CO_2-e ha^{-1} year^{-1} cm s$^{(1}$ (0.05 t C ha^{-1} year^{-1}); the average across all sites were the grazing exclusion treatment was maintained. While these rates of sequestration are modest at the hectare scale, they equate to a substantial potential soil C storage when extrapolated across the mulga lands, or even across individual properties, which are frequently very large. Based on these estimates of C sequestration, if 50% of eastern Australia's mulga lands bioregion (25.4 million ha) were managed for biosequestration and biodiversity through the control of all herbivores, then approximately 2.3 Mt CO_2-e year^{-1} could be sequestered in the soil alone which equates to approximately 0.42% of Australia's total emissions which were approximately 550 Mt CO_2-e year^{-1} in 2008 (Department of Climate Change and Energy Efficiency, 2010). It should be noted that these rates are calculated on the basis of the potentially biased, low bulk density measurements made at the sites. Thus, these estimates of C sequestered may be considered conservative. However, estimates from this study are lower than the soil C sequestration rates reported by Schuman et al. (2002) for semi-arid lands of the United States.

This study has also shown that above ground woody biomass increases with the exclusion of grazing due to the regeneration of mulga; but the response is highly variable across the study sites. If soil C and above ground living and dead woody biomass are combined, the study indicates that a three to four-fold increase (over soil C values only) to between 0.92 and 1.1 t CO_2-e ha^{-1} year^{-1} can be sequestered with the successful removal of all grazing pressure (including domestic, feral and native herbivores). Carbon sequestration potential for 50% of the approximately 25.4 million hectares of the mulga lands bioregion, for both soil and above ground accumulations, could range between 11.6 and 14 Mt CO_2-e year^{-1}, which is between 2.12 and 2.54% of Australia's total emissions.

Recently, the CSIRO (2009) assessed greenhouse gas sequestration and mitigation potential of rural land use change in Queensland, and also reviewed the Garnaut Report (Garnaut, 2008). From the thirteen broad options studied by CSIRO, they identified forestry activities (such as C forestry) as the best option as it was forecast to sequester 105 Mt CO_2-e year^{-1} and is easily implemented. The Wentworth Group of Concerned Scientists (2009) addressed the previous publications and acknowledged the power of terrestrial C to contribute to Australia's climate change solutions. They encouraged Australia to use the new C economy to address environmental challenges such as repairing degraded landscapes, restoring river corridors and conserving biodiversity.

This study suggests that the potential for the mulga lands of eastern Australia (most of which occur in south west Queensland) to sequester C, is between the estimates provided by Garnaut (2008) and CSIRO (2009). Thus, these semi-arid mulga systems do have the potential to contribute to Australia's mitigation efforts that could be as high as 2 or 2.5% of total annual emissions. What the study has also identified is important ecological changes associated with grazing removal which have significant hydrological and biodiversity benefits.

In this study, the assessment of C accumulation was limited to the surface 30 cm. It is possible that biological activity extends deeper within the exclosures than outside, based on the observations of greater infiltration, and hence increased available water and herbaceous cover. However, change in organic C content at depth is likely to be modest. Dalal et al. (2005) in a paired site comparison of the C contents of a soil under mulga, or cleared and sown to buffel (*Cenchrus ciliaris*) pasture for 20 years, reported that significantly greater C contents in the mulga soil were limited to the 0–5 cm layer, and that all indication of difference had disappeared by 30 cm. Thus the 30 cm sampling depth used in this study is considered sufficient to capture the majority of the change in soil C content, but is not sufficient to characterize the entire soil C store.

It should be noted that the small core sampling method used in this study has been reported to result in the underestimation of bulk density in rocky, and very hard soils (Flint and Childs, 1984; Andraski, 1991), and in this study resulted in some particularly low values (<1.0 g cm$^{(3}$) which were discarded. Nevertheless, the average bulk density values obtained in this study are much lower than typically reported for these soils. For example, Dalal et al. (2005) reported bulk densities of 1.4–1.5 for a Kandosol under mulga and buffel grass pasture, while Green (1992) reported bulk density of 1.3–1.4 for grazed mulga woodland. The sampling method used in this study is likely to have introduced a systematic bias, underestimating the bulk density of the soils. Furthermore, the use of these low bulk density measurements in the calculation of C stocks per unit area, will result in an underestimate of the C pool, and hence in an underestimate (conservative estimate) of C sequestration achieved by grazing exclusion.

4.2. Environmental effects of grazing exclusion

Ecologically, the most important findings of this study are the observations relating to ecosystem processes and function, especially water infiltration rates, water availability, and herbaceous cover.

The long-term exclusion of stock has resulted in significant ecological changes, particularly in those sites where exclusion is greater than three decades. For many sites the obvious visual changes include the significant regeneration of mulga and an increase in the herbaceous cover within the exclosures. These vegetation changes, while important, are not a surprising finding, as continuous grazing pressure results in reduced herbage cover and can also affect the recruitment of edible shrubs and trees such as mulga. For example, herbaceous cover inside the exclosures averaged over 40% while the corresponding value outside was approximately half of this (21.7%). It is also obvious therefore that above ground C (in trees) increases as a consequence of stock exclusion. For living trees of less than 15 cm diameter, which are assumed to have grown since the establishment of most of the exclosures, the standing crop of above ground C was estimated at 5.3 t ha^{-1}, while outside the exclosures it was only 1.1 t ha^{-1}. However, the competition created by a large cohort of reasonably densely distributed mulga plants does appear to affect the capacity of large/mature trees to withstand extended periods of below average rainfall (see Tables 5 and 6) the living and dead comparisons in Section 3.

Saturated hydraulic conductivity, and hence infiltration rates, were significantly higher inside the exclosures, and this is reflected in the greatly increased water availability within the enclosures at sites sampled following rain. These two key variables indicate the restoration of ecosystem processes at the site scale. Because the soil is able to capture, cycle and effectively retain water for primary productivity at small scales, 'leakage' of resources is minimised (cf. Tongway and Hindley, 2004). If scaled up to the landscape/property the environmental benefits of grazing reduction for restoration that results from this conservation of resources are likely to be significantly greater than can be measured at the exclosure scale.

The major changes in functionality (infiltration rates and water availability) mean that in a semi-arid environment small inputs of rainfall are much more efficiently retained and utilised at small scales, this in turn allows for germination, growth, and successful regeneration of species that are at a competitive disadvantage under continuous grazing pressure exerted by domestic, feral and native herbivores. The consequences of this, as indicated by the numbers of grass species only found inside the exclosures, suggests that if larger areas of land could be effectively controlled for grazing pressure, significant biodiversity benefits could accrue while simultaneously sequestering C in soil and biomass.

5. Conclusions

This study demonstrates that managing the landscape for its carbon sequestration potential is likely to have a significant positive impact on soil physical and chemical properties improve ecosystem functioning and biodiversity in areas semi-arid parts of Australia where birds and meso-mammals have suffered catastrophic decline, which is indicative of the biodiversity decline is such regions. These benefits can be greatly increased and community benefits increased with the adoption of more active interventionist management approaches.

Restorative management through the control of total grazing pressure in mulga lands of eastern Australia has the potential to sequester C in soils and above ground biomass at rates ranging between 0.91 and 1.1 t CO_2-e ha^{-1} year^{-1} over a period of between 30 and 40 years. Reduction of grazing pressure could be achieved through the removal of stock, and the control of artificial watering points to reduce populations of native and feral grazing animals.

The implementation of a strategy that involved existing land holders, and included restorative management systems, is a policy setting that would improve the economic wellbeing of the communities of the region, aid biodiversity recovery, and provide a managed C sink. What is required, are the appropriate policy settings to encourage this change.

This study is a 'passive' experiment that has exploited existing exclosures for a purpose far removed from that intended for them. This 'serendipitous' experiment has not unexpectedly produced results that take much longer to show, and the small scale of the exclosures used in the study may also act to reduce rates of change. The extension of active management to new active restoration strategies incorporating grazing pressure management and property planning, is potentially viable and could increase both the C and biodiversity rates of change considerably, relative to those predicted in this study.

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