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Plant and soil surface responses to a combination of shrub removal and grazing in a shrub-encroached woodland

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A B S T R A C T

Shrub encroachment into open woodland is a widespread phenomenon in semi-arid woodlands worldwide. Encroachment or woody thickening, is thought to result from overgrazing, changes in fire regimes and increased atmospheric carbon dioxide concentrations. Eighteen years after one-off shrub removal by ploughing we assessed the effects of four different land management systems resulting from two levels each of grazing (grazed, ungrazed) with and without ploughing, on the cover of landscape units, soil surface condition, diversity of understorey plants and density of shrubs. We recorded 2–7 times more patches under conventional conservation (unploughed-ungrazed) than the others treatments, and plant cover and diversity were greater on the two conservation (ungrazed) plots, irrespective of ploughing. Soils under shrubs and log mounds had greater indices of infiltration, stability and nutrients. Shrub density under the active pastoral (ploughed-grazed) treatment was two and a half times greater than that in other treatments, but results were not significant. The effects of different treatments on shrubs were largely species-specific. Overall, our results suggest that ploughing does not provide long-term control of encroaching shrubs.

1. Introduction

The encroachment of native woody plants into former open grassland or open woodland is an increasing global phenomenon that has been likened to the spread of exotic plants (Noble, 1997; Van Auken, 2000; Archer et al., 2001; Maestre et al., 2009). The encroachment of snakeweed (*Gutierrezia* spp.) (Sterling et al., 1999), mesquite (*Prosopis glandulosa*) and creosote bush (*Larrea tridentata*) in western United States rangelands (Buffington and Herbel, 1965), and turpentine (*Eremophila sturtii*), budda (*Eremophila mitchellii*) and narrow-leaf hopbush (*Dodonaea viscosa* var. *angustissima*) in eastern Australian rangelands (Hodgkinson, 1979) are examples of native species that exhibit some invasive qualities. The causes of this phenomenon, variously referred to as shrub invasion, encroachment or woody thickening, are varied, and appear to be due to the interaction of several factors (Archer et al., 1995). Changes in land-use practices from frequent wildfire to controlled, infrequent fire are exacerbated by increases in stocking rates of both domestic and native animals over large areas of rangelands (Wilson, 1990; Harrington, 1991; Archer et al., 1995), which have reduced fire fuel loads. Variation in rainfall (Fensham et al., 2005) and increases in global CO_2 concentrations are thought to have benefited C_3 woody species at the expense of C_4 grasses (Polley et al., 1993; Chapin et al., 2002), and studies within long-ungrazed exclosures suggest that climate change may be responsible, at least in part, for some changes in shrubland communities (Grover and Musick, 1990).

The generally negative perception of native shrubs as invaders is highly context-dependent (Colautti and MacIsaac, 2004) and based largely on the effects that shrubs have on reducing pastoral productivity in agricultural, production-based landscapes. It is unclear, however, whether shrubs *per se* or interactions between grazing and climate are responsible for the putative reductions in pastoral production in encroached woodlands and grasslands. Woody shrub encroachment into grasslands is known to alter the balance between runoff and infiltration (Rostagno et al., 1991) and transpiration and evaporation (Aguiar et al., 1996), affect the spatial

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distribution of soil nutrient pools (Schlesinger et al., 1996) and produce far reaching effects on ecosystems. These putative negative effects are, however, not universal. Shrubs are important for moderating a range of ecosystem services because they influence micro-climate, reduce erosion and nutrient loss, increase water holding capacity, maintain soil structure and stability, and provide habitat for a range of taxa (Maestre et al., 2009). Shrubs intercept solar radiation, lowering soil surface temperatures, reducing evapotranspiration and increasing soil moisture contents (Vetaas, 1992). Shrubs also regulate pathways of nutrient flow by acting as nutrient pumps (Vetaas, 1992). Recovery rates of heavily grazed systems are faster with the presence of shrubs than with perennial grasses alone (Mazzarino and Bertiller, 1999). Shrubs are not only more resistant to drought (Richmond and Chinnock, 1994; Booth et al., 1996), but they can also tolerate extremes of fire, salinity and frost (Richmond and Chinnock, 1994) and under grazed conditions, have a competitive advantage over perennial grasses (Maconochie, 1982; Booth et al., 1996). Shrub encroachment in Mediterranean systems has recently been shown to be associated with improved ecosystem function (Maestre et al., 2009).

There have been many attempts to transform shrublands back to their putative original grassland-woodland matrix through programs of destocking, exclosure, or by using mechanical, biological and chemical techniques. Stock removal through exclosure produces conflicting results (Trodd and Dougill, 1998; Angassa and Oba, 2007) and in most cases, shrubs continue to maintain their dominance even after complete grazing removal (West et al., 1984; Eccard et al., 2000; Valone et al., 2002; Guo, 2004; Mengitsu et al., 2005; Angassa and Oba, 2007). Mechanical removal of shrubs generally produces only short-term changes in plant community structure (Morton and Melgoza, 1991; Robson, 1995; Allegretti et al., 1997), often stimulates shrub regrowth, and leads to the eventual persistence and dominance by woody shrubs (e.g., Ruthven III et al., 1993). Post-treatment management of stocking rates is rarely considered by land managers, and many failures are thought to be due to the suppression of grass growth due to overgrazing (Cox et al., 1984). Similarly, landscape-level processes often overwhelm small-scale shrub manipulation treatments, limiting the success of many restoration programs (Herrick et al., 2006).

Here we consider the effects of four land management treatments on the structure and composition of a shrub-encroached woodland. Our study site is typical of encroached woodlands in eastern Australia, which are characterised by dense native shrubs, sparse perennial grasses, and an extensive cover of bare, eroded soil. The four treatments represent different land management scenarios that reflect how managers in eastern Australia manage shrubs within the context of both pastoral production and conservation. The 'conventional pastoral' option involves set stocking and no shrub removal, while the 'active pastoral' treatment involves the removal of shrubs followed by the maintenance of current grazing practices. 'Conventional conservation' is a strategy practiced by managers of national parks and conservation reserves whereby livestock are removed but shrubs are not controlled. 'Active conservation' involves the removal of livestock and some shrub removal in order to meet some specific conservation objectives such as altered habitat for key plant or animal species (Table 1). While most of the semi-arid grazing land in eastern Australia falls within the conventional pastoral scenario, increasingly larger areas of shrubland and woodland are being 'treated' (shrub removal) through incentives programs driven by community-based land management agencies such as Catchment Management Authorities (http://cw.cma.nsw.gov.au) under a scenario of continued livestock grazing. Apart from localised removal of shrubs to improve management of conservation reserves (e.g., removal along fences and tracks), we are unaware of any large scale 'active conservation' in the context of physical removal of shrubs.

Table 1
Summary of the four ploughing and grazing treatment scenarios and their extend in eastern Australia.

Scenario	Ploughed	Grazed	Extent	Disturbance intensity
Conventional conservation	No	No	Localised (<10% of landholders)	Low
Active conservation	Yes	No	Very uncommon (<1% of landholders)	Moderate
Conventional pastoral	No	Yes	Widespread (>50% of landholders)	Moderate
Active pastoral	Yes	Yes	Very common (25–50% of landholders)	High

We used a long-term exclosure experiment to test three predictions about the response of soil surfaces, groundstorey vegetation and shrubs along a gradient in disturbance resulting from a one-off shrub removal by blade ploughing (*syn*. root ploughing) followed by either grazing or exclosure. Blade ploughing has been promoted widely by agencies in eastern Australia since the late 1970s for control of shrubs (Robson, 1995), though anecdotal evidence suggests that it provides only short-term shrub control. A long-term perspective is needed to adequately evaluate the response of shrubs and soil surfaces to disturbance. Because many studies have considered only short-term responses (e.g., Robson, 1995), our study adds substantially to society's understanding of shrub control in semi-arid woodlands. Second, our study combines two elements of the encroachment problem; removal of the shrubs themselves, and the interactive effect of grazing. Without controlling for the grazing effect, it is difficult to gauge the long-term consequences of shrub removal within a pastoral context.

Grazing is known to reduce the size and number of vegetated patches (e.g., van de Koppel et al., 2002) and thus we predicted a marked decline in measures of landscape function such as soil surface condition and the number and size of resource-accumulating patches with increasing disturbance i.e., from conventional conservation to the actively-managed, pastoral treatment (Table 1). We also predicted that grazing and ploughing would reduce plant diversity and composition by maintaining the dominance by shrubs, consistent with results worldwide (e.g., Bisigato and Bertiller, 1997; Ruthven III et al., 1993). Third, we anticipated shrub species-dependent treatment effects, specifically increases in the densities of grazing- and ploughing-tolerant species such as *E. sturtii* and *Eremophila gilesii* but reductions in grazing- and ploughing-sensitive species such as *Acacia aneura* and *D. viscosa* under the active pastoral treatment. We expected species that are both ploughing-tolerant and grazing-intolerant, such as *Eremophila longifolia*, to be found in the active pastoral treatment while the conventional conservation treatment would be expected to be characterised by a mixture of intolerant species as well as grazing- and ploughing-tolerant species.

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), which is dominated by mulga (*A. aneura*) woodlands. The landscape is characterised by sandplain with low west-east trending sandy rises and dunes of Quaternary aeolian alluvium. Minor to moderate windsheeting and water-sheetings are common. The sandplain unit is level to slightly undulating (slope < 1%) and the soils are dominated by sandy red earths with sandy loam to loam texture. Mean annual rainfall at Wapweelah is about 312 mm, and about 45% more rain falls during the six warmer months than the cooler months (Robson, 1995).

The sandplain unit of Gumbalie land system is characterised by dense shrubs dominated by turpentine, narrow-leaf hopbush, green turkey bush (*E. gilesii*), budda and wild orange (*Capparis mitchellii*). There are few scattered trees of mulga, ironwood (*Acacia excelsa*), bimble box (*Eucalyptus populnea*), belah (*Casuarina cristata*), rosewood (*Alectryon oleifolius*), leopardwood (*Flindersia maculosa*), supplejack (*Ventilago viminalis*), wilga (*Geijera parviflora*), gidgee (*Acacia cambagei*) and brigalow (*Acacia harpophylla*). The understorey layer is sparse, and at the time of observation consisted of heavily grazed wollybutt (*Eragrostis eriopoda*), mitchell grass (*Thyridolepsis mitchelliana*), mulga oats (*Monochather paradoxa*), speargrass (*Austrostipa scabra*), other grasses and forbs. This land system is typical of areas targeted for blade ploughing (Walker, 1991; Robson, 1995). Shrubs do not form a community resembling the 'brousse tigre' vegetation of southern Africa but grow in isolated patches, sometimes forming islands. Many shrubs occupy bare area resulting from the loss of perennial grasses through overgrazing (Daly and Hodgkinson, 1996).

2.2. Shrub removal design

In 1990, three replicate blocks of 200 m × 400 m were established, and each divided into four equal plots of 100 m wide by 200 m long. The two central plots were then enclosed in a 6 m high, herbivore-proof fence. Half of both the fenced and unfenced plots were then ploughed and the other half unploughed. The grazed plots were subjected to grazing by sheep, goats, cattle, kangaroos, rabbits and camels. The three replicate blocks varied from 1 km to 5 km apart.

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. The cutting depth was maintained at 20–30 cm to sever the taproots of *Eremophila* spp. in order to achieve a maximum kill rate (Wiedemann and Kelly, 2001). Examination of fallen shrubs confirmed a satisfactory ploughing effect over most of the treated area (Robson, 1995).

2.3. Assessment of landscape units

In August 2008, 18 years after ploughing and fencing, we established a 50 m transect through the centre of each plot in order to measure the composition of different landscape units (i.e., plains, shrub hummocks, and log mounds) in each treatment. Shrub hummocks are the elevated accretions around woody plants and log mounds are accumulations of wind and water eroded material deposited against ploughed shrubs and fallen timber. Along with shrub hummocks, these are sufficient to impede sediment, litter and seed movement. The intervening area between shrub hummocks and/or log mounds, comprised plains, generally bare, but sometimes with a poorly developed cover of cryptogamic crusts.

2.4. Landscape-scale functional measures

The same 50 m transect was also used to assess the functional integrity of each plot in terms of the degree to which resources such as runoff water, entrained sediment, seeds, nutrients and organic matter are retained within patches such as grass tussocks, which occur in the interspaces on the plains. Landscape function analysis (LFA) has been shown to be strongly related to empirical measures of soil function (e.g., Ludwig and Tongway, 1995; Maestre and Cortina, 2004). Along this transect we recorded the total length and width of resource-accumulating patches such as grass tussocks, logs, tree and shrub hummocks, and the distances between the these units (fetches), which represent resource shedding. A detailed assessment of the LFA technique is given in Ludwig and Tongway (1995).

2.5. Soil surface morphology

Detailed measurements of the morphology of the soil surface were made within the landscape units identified on each plot using quadrat-based methods according to the methods of Tongway (1995). Thirteen soil surface features were assessed within ten 0.5 m^2 quadrats, stratified according to the relative cover of different landscape units on a plot. These surface features were: 1) surface stability, which measures the ease of soil particles to be removed by wetting and the ability of surface or projected plant cover to protect them; 2) soil surface roughness, which assesses how rough the soil surface is and the level to which it will act as micro-depression for mobile resources; 3) surface cracking, which measures the proportion of the surface covered with cracks, and the potential for the surface to be dislodged by erosion; 4) cryptogam cover which determines the cover of non-vascular plants on the surface; 5) crust brokenness, which assesses to what extend the surface crust is broken and the degree to which raindrop impact will deform the soil surface; 6) type and severity of erosion, where forms of erosion were used to determine the severity of wind and/or water erosion; 7) litter cover; 8) whether litter is derived from local plants or transported from elsewhere; 9) the degree to which litter is decomposed and incorporated into the soil, 10) perennial vegetation cover, which measures the basal and canopy cover; 11) deposited material which measures the nature and amount of soil and litter transported and deposited on the query zone; 12) surface stability to rapid wetting, using a simple slake test; and 13) soil surface texture, which indicates how good soil infiltration is and its capacity to store water and support plants.

The 13 soil surface attributes described above were used to derive three biogeochemical indices which describe the extent to which the soil: 1) cycles nutrients (nutrients), 2) captures water (infiltration) and 3) resists erosion (stability) (Tongway, 1995). For a particular quadrat the value of each attribute was given a score, usually from 1 to 5, with a larger score equating with a healthier functional surface. Thus for example, the soil surface roughness classes of <5 mm, 5–8 mm, 8–15 mm, 15–25 mm and >25 mm were assigned scores of 1, 2, 3, 4 and 5 respectively such that a rougher surface, which implies a better soil condition, receives a higher score. The overall index for stability for a given quadrat was derived as the sum of the seven scores for surface cracking, surface stability, crust brokenness, degree of erosion, cryptogamic cover, vegetation cover and litter cover expressed as a percentage of 40, the maximum possible score. In a similar manner, four of the surface attributes described above (surface roughness, cryptogamic cover, vegetation cover, and litter cover) were used to derive a score for the nutrient status of the soil, which is based on the ability of the soil to cycle and retain nutrients. Finally, values for vegetation cover, a combined product of the three litter scores, surface roughness, surface stability and soil texture were used to derive an index of infiltration (Tongway, 1995).

2.6. Plant cover and diversity

We recorded separately, the height classes of all dead and live shrubs, by species, along one 100 m × 2 m transect on each plot.

Table 2
Landscape unit-level mean (±SE) values for measures of soil surface condition for the three landscape units. Letters within a row indicate a significant difference in the attribute at $P < 0.05$.

Soil surface indices	Shrub hummock		Log mound		Plain	
	Mean	SE	Mean	SE	Mean	SE
Nutrient index (%)	24.1^a	1.2	22.4^a	1.4	16.9^b	1.2
Infiltration index (%)	32.7^a	1.8	33.2^a	2.1	24.5^b	1.8
Stability index (%)	58.9^a	1.7	54.8^b	2.0	50.8^b	1.7

Shrubs were measured on a scale of 1 < 0.25 m, 2 = 0.25–0.50 m, and thereafter at 0.50 m increments, to 10 = 3.50–4.00 m. Additionally, within a 20 m × 20 m quadrat centrally located along the 50 m transect, we used a quadrat of 100 grid points to estimate the cover of bare ground, plant, litter and cryptogamic crusts and the cover of all groundstorey, midstorey and upperstorey vascular plant species.

2.7. Statistical analyses

We used a mixed-models ANOVA with two strata to examine differences in the three soil surface indices (infiltration, stability, nutrient) in relation to the four treatments and three landscape units. The first stratum considered treatment, and the second stratum landscape unit and its interaction with treatment. To examine differences in shrub density and landscape functions, including the number of resource-accumulating patches, fetch length and patch width, we used a randomized block ANOVA after scaling the data up to the landscape level by adjusting for the relative cover of different landscape units at each plot. Data were checked for normality and homogeneity of variance (Levene's test) using Minitab version 15 prior to analyses. *Post-hoc* differences in means were tested using Least Significant Difference testing. Differences in shrub height class distributions in relation to the treatments were determined using Ordinal Logistic Regression using Genstat version 12.

Permutational multi-variate analysis of variance (PERMANOVA, Anderson et al., 2008) with the Bray–Curtis similarity coefficients was used to test whether the composition of 1) groundstorey plants and 2) shrubs differed in relation to the four treatments. Values for richness, evenness, abundance (number of individuals) and number of species were determined using the DIVERSE function of PERMANOVA and differences in relation to treatment assessed using the same ANOVA structure described above for the landscape-level study. The SIMPER subroutine of PERMANOVA was used to determine which species contributed most to the dissimilarity among treatments. The degree of association of different species with the four treatments was assessed with Indicator-Species Analysis using PC-ORD (McCune and Mefford, 1999). Indicator values (IV) are maximal (100%) when all individuals of a given species are restricted, for example, to a given treatment, or all samples from that treatment contain that species.

3. Results

3.1. Landscape unit-level changes

At the scale of landscape units, plains always had the lowest indices of nutrients ($F_{2,16} = 16.9$, $P = 0.003$) and infiltration ($F_{2,16} = 24.5$, $P < 0.001$) compared with shrub hummocks and log mounds, except for the index of stability where the difference was only significant between plains and shrub hummocks ($F_{2,16} = 50.8$, $P = 0.002$) (Table 2).

We did not find any significant interactions, however, between landscape unit and treatment, indicating consistent effects of landscape unit among the treatments. We only found one significant treatment effect, with active conservation plots having 1.2 times greater values of the nutrient index than active pastoral plots ($F_{2,16} = 7.09$, $P = 0.012$).

3.2. Landscape-level changes

At the landscape scale, we detected no significant differences in the nutrient ($P = 0.18$), infiltration ($P = 0.16$) or stability ($P = 0.69$) indices among the four treatments (Table 3). However, trends for these indices were similar, with a general decline from the least disturbed to the most disturbed treatments (i.e., from conventional conservation to active pastoral).

There was a significant treatment effect, however, on the number of resource-accumulating patches per 50 m of transect, with two- to seven-times more patches in the conventional conservation treatment than the others ($F_{3,8} = 27.0$, $P < 0.001$, Table 3). Mean distance between patches (fetch length) also varied significantly in relation to different combinations of grazing and ploughing, with fourfold larger fetches in the conventional pastoral plots (mean = 20.72 m) than the conventional conservation plots (4.77 m; $F_{3,8} = 5.91$, $P = 0.02$). There were no treatment effects, however, on patch width ($P = 0.35$; Table 3).

3.3. Plant, litter and cryptogam cover

Plant cover was greatest under the active conservation treatment (43.3 ± 4.4%; mean ± SE) compared with the other management scenarios (20.3–27.3%; $F_{3,8} = 6.17$, $P = 0.018$) but there was no difference in cover of bare ground ($P = 0.35$) nor litter cover ($P = 0.56$). Cryptogam cover was about four-times greater under the conventional (unploughed) treatments than the active management (ploughed) treatments ($F_{3,8} = 2.56$, $P = 0.043$; Table 4).

Table 3
Mean (±SE) values for measures of landscape function for the four grazing and ploughing treatments. Letters within a row indicate a significant difference in the attribute at $P < 0.05$.

Attribute	Conventional conservation		Active conservation		Conventional pastoral		Active pastoral	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Nutrient index (%)	20.1^a	1.6	19.7^a	0.6	16.9^a	2.9	14.7^a	1.1
Infiltration index (%)	29.9^a	3.9	25.8^a	1.1	22.2^a	2.3	22.3^a	1.5
Stability index (%)	54.6^a	1.9	50.3^a	5.7	49.5^a	6.4	46.9^a	2.4
No. of resource-accumulating patches[1]	9.3^a	0.3	3.3^b	0.9	1.3^b	0.3	3.3^b	0.9
Fetch length (cm)	447.0^a	15.2	1240.0^{ab}	240.0	2072.0^b	414.0	1243.0^{ab}	240.0
Patch width (cm)	44.3^a	12.0	32.7^a	5.6	35.3^a	18.7	13.8^a	1.6

Table 4
Mean (±SE) values of plant, litter, bare ground and cryptogam cover, and measures of community composition for groundstorey and midstorey species for the four treatments. Letters within a row indicate a significant difference in the attribute at $P < 0.05$.

Attribute	Conventional conservation		Active conservation		Conventional pastoral		Active pastoral	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Plant cover (%)	25.0[a]	2.9	43.3[b]	4.4	20.5[a]	5.0	27.3[a]	3.4
Litter cover (%)	11.7[a]	3.3	11.0[a]	2.1	7.9[a]	0.3	9.3[a]	1.3
Bare ground (%)	33.3[a]	7.3	36.3[a]	7.9	43.6[a]	14.7	57.3[a]	5.8
Cryptogam cover (%)	30.0[a]	8.7	9.3[b]	4.3	28.0[a]	17.4	6.0[b]	4.5
No. of species	20.3[ab]	1.7	25.0[a]	2.0	15.3[b]	1.5	15.0[b]	3.1
Abundance	13.3[ac]	1.7	18.0[bc]	1.2	9.7[a]	1.7	12.0[ac]	1.0
Species richness	7.6[a]	1.0	8.3[a]	0.7	6.4[a]	0.4	5.6[a]	1.0
Species evenness	0.9[a]	0.01	0.9[a]	0.01	0.9[a]	0.03	0.9[a]	0.02
Dead shrubs (ha^{-1})	1516.0[a]	498.0	1733.0[a]	833.0	700.0[a]	150.0	2050.0[a]	828.0
Live shrubs (ha^{-1})	3133.0[a]	421.0	3400.0[a]	1505.0	2300.0[a]	1404	5750.0[a]	3404.0

3.4. Plant community structure and composition

Averaged over plots, the active conservation treatment had about 60% more species than the active pastoral treatment ($F_{3,8} = 4.92$, $P = 0.032$). The active conservation treatment also supported the greatest number of individuals (18.0 ± 1.15) while the conventional pastoral plots had the least (9.7 ± 1.67; $F_{3,8} = 5.76$, $P = 0.021$). There were no significant effects on species richness ($P = 0.16$) nor evenness ($P = 0.71$) (Table 4).

The composition of understorey plants varied significantly in relation to the different treatments (pseudo $F_{3,8} = 1.73$, P (perm) = 0.027). There was a slight difference in the complement of species between the extreme treatments (active pastoral and conventional conservation: $t = 1.98$, P (perm) = 0.064). Four species, two forbs (*Sclerolaena divaricata, Sclerolaena convexula*) and two perennial grasses (*E. eriopoda*, *Monachather paradoxa*) explained more than 50% of the difference between active pastoral and conventional conservation. The two *Sclerolaena* spp. were more abundant in the active pastoral plots while the two grasses were more abundant in the conventional conservation plots. *Maireana appressa* (IV = 84.2, $P = 0.053$) was greatest in conventionally-conserved plot compared to other plots.

3.5. Shrub community structure and composition

There were no significant differences in the total densities of either live or total (live + dead) shrubs in relation to the four treatments ($P = 0.62$). There were, however, about 2.5-times more shrubs in the active pastoral treatment (5750 ± 3404 shrubs ha^{-1}) than under conventional pastoral (2300 ± 1404 shrubs ha^{-1}; Table 4).

The distribution of shrub sizes on the most disturbed (active pastoral) treatment was skewed towards a large number of small individuals, while the least disturbed treatment (conventional conservation) showed a more hump-shaped size class distribution (Fig. 1). Grazed plots (both under active and conventional pastoral) had a greater proportion of juvenile shrubs than ungrazed (conserved) plots ($F_{3,21} = 13.6$, $P < 0.001$; Fig. 1), which was due principally to the presence of large cohorts of *E. gilesii* seedlings (Fig. 2).

Ploughing and grazing treatments favoured or hampered shrub species to a different degree. *E. sturtii* ($F_{3,8} = 5.26$, $P = 0.027$) was almost 100-times denser under conventional conservation than under active conservation (Table 5, Fig. 2). *E. longifolia* shrubs showed the opposite trend, with 8-times greater density at active conservation plots than conventional conservation plots

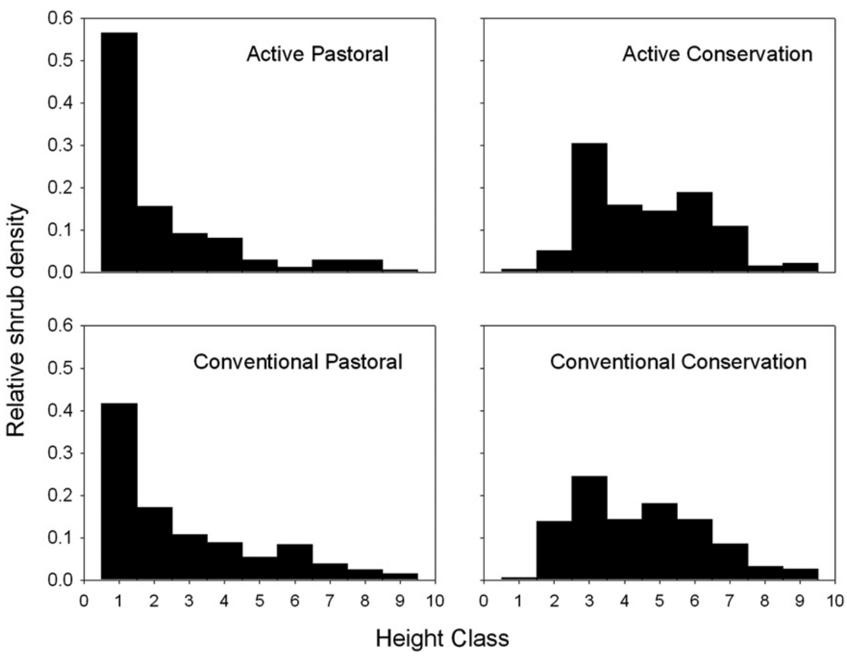

Fig. 1. Distribution of shrub heights for all shrubs in relation to the four grazing and ploughing treatments. Class intervals range from 1 = ≤ 0.25 m to 10 = 3.50–4.00 m (see Methods).

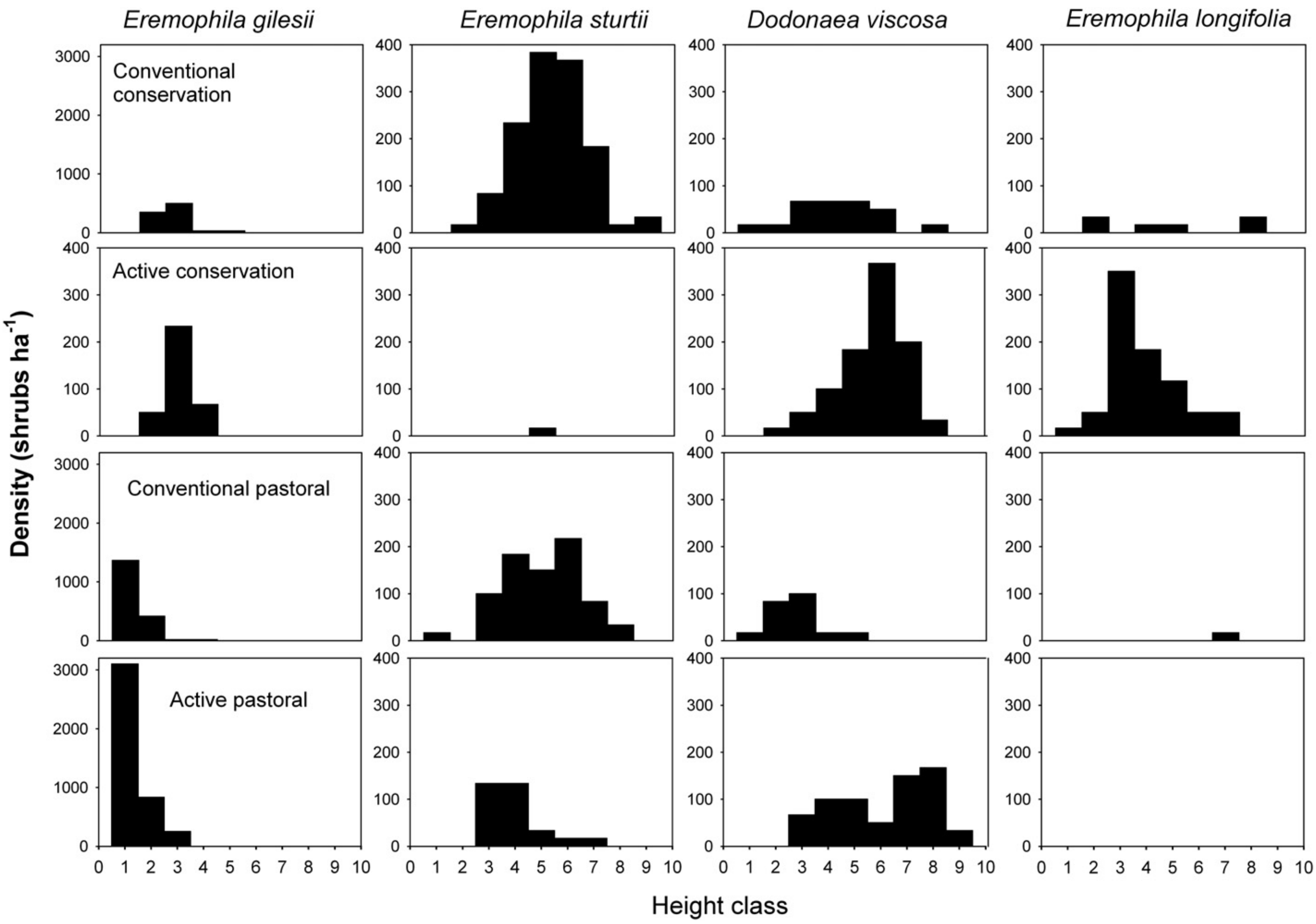

Fig. 2. Distribution of shrub heights for *Eremophila gilesii, Eremophila sturtii, Dodonaea viscosa* and *Eremophila longifolia* by density (ha^{-1}) for the four grazing and ploughing treatments. Class intervals range from 1= ≤ 0.25 m to 10 = 3.50–4.00 m (see Methods). Conv. = conventional, conserv. = conservation. Note different scale of *y*-axis for *Eremophila gilesii*.

($F_{3,8} = 8.20$, $P = 0.008$). *E. longifolia* shrubs were not found at active pastoral plots (Table 5; Fig. 2).

There were no treatment effects on the number of shrub species ($P = 0.18$), the number of individual shrubs ($P = 0.14$), shrub species richness ($P = 0.20$) nor evenness ($P = 0.39$). There were, however, significant differences in shrub composition between treatments (pseudo $F_{3,8} = 2.27$, P (perm) = 0.02). Differences were generally due to a dominance of *A. excelsa* in the conventional conservation plots compared with the active pastoral plots, more *E. sturtii* in the conventional pastoral and conservation plots than the active conservation plots, and more *E. longifolia* in the active conservation plots than the active pastoral plots (Table 5). *E. longifolia* (IV = 94.7, $P = 0.006$) and *A. aneura* (IV = 94.7, $P = 0.006$) were strong indicators of active conservation and conventional conservation plots, respectively (Table 5). SIMPER analyses indicated that ploughed (active pastoral and active conservation) plots supported more small (<0.50 m) *E. gilesii* and moderate to large-sized (>1.00 m) *E. sturtii* than unploughed plots, while ploughed plots supported more large (>1.50 m) *D. viscosa* than unploughed plots (Fig. 2).

4. Discussion

4.1. Treatment effects on landscape units and surface condition

Both the density and size of resource-accumulating patches declined with increased disturbance, consistent with our prediction and with observations worldwide of substantial effects of grazing and disturbance on vegetation patches (Roundy and Jordan, 1988; Allegretti et al., 1997; Eccard et al., 2000; van de Koppel et al., 2002; Golodets and Boeken, 2006). The least disturbed treatment (conventional conservation) had the greatest number of patches (9.3 per 50 m of transect), with average microrelief of 5–10 cm high. Natural rates of patch formation are probably low in the semi-arid woodlands, therefore it is not surprising that we failed to detect a recovery in their density 18 years after ploughing, even under exclosure (active conservation). In a similar landscape, artificial patches created by placing mulga (*A. aneura*) branches on the soil surface accreted at a rate of about 0.5–0.7 mm yr^{-1} compared with an erosion rate of 0.8–0.9 mm yr^{-1} in areas devoid of these structures (Tongway and Ludwig, 1996). Based on Tongway and Ludwig (1996) accretion rates, recovery of patches at our study site would take more than a century under exclosure or about 160 years under grazing in the shrubby woodlands. These projections highlight the fact that landscape change in semi-arid woodlands is protracted (Coffin et al., 1996; Valone et al., 2002; Guo, 2004), and that 18 years of recovery is insufficient to restore landscape structure.

Though the density of patches declined, we did not detect significant changes in bare soil nor any treatment effects on the three surface health indices. Cryptogam cover was, however, lower under the active (ploughed) treatments, irrespective of grazing status. Reduced cryptogam cover is a typical responses to increased

Table 5
Mean (±SE) density of shrubs (ha^{-1}) in relation to the four grazing and ploughing treatments. Differences were significant only for *Eremophila longifolia* and *E. sturtii*.

Species	Conventional conservation		Active conservation		Conventional pastoral		Active pastoral	
	Mean	SE	Mean	SE	Mean	SE	Mean	SE
Acacia aneura	100	50	50	29	0	0	17	17
Acacia burkittii	100	100	0	0	0	0	0	0
Acacia excelsa	133	33	17	17	167	117	0	0
Acacia tetragonophylla	17	17	0	0	0	0	0	0
Atalaya hemiglauca	0	0	17	17	0	0	0	0
Alectryon oleifolius	0	0	0	0	33	33	0	0
Dodonaea viscosa subsp. *angustissima*	533	324	1317	825	283	192	750	603
Eremophila desertii	67	44	33	33	183	136	133	83
Eremophila gilesii	1733	773	583	192	2800	2800	6083	6008
Eremophila glabra	33	33	67	44	67	67	150	150
Eremophila longifolia	100[a]	76	817[b]	262	17[c]	17	0[c]	0
Eremophila mitchellii	17	17	0	0	167	167	267	159
Eremophila sturtii	1683[a]	232	17[b]	17	1400[a]	633	383[c]	169
Grevillea striata	33	33	67	33	0	0	0	0
Pimelea microcephala subsp. *microcephala*	33	33	0	0	17	17	0	0

disturbance in semi-arid environments (Anderson et al., 1982; West, 1990), and while direct physical disturbance by ploughing would have been responsible for the initial crust reduction at our plots, trampling by stock would compromise its recovery (Williams et al., 2008), resulting in persistent soil and nutrient loss for many decades (Neff et al., 2005). Crust cover is sparse to moderate in the semi-arid wooded rangelands (Eldridge, 2001) and any exacerbation by grazing on an already disturbed surface will likely retard the development of the crust, and could promote a crust dominated by cyanobacteria (Eldridge and Greene, 1994).

Water-repellent crusts enhance the redistribution of water from the interspaces, thereby augmenting water supply to the patches. Breakdown in crust continuity by animal trampling alters runoff-infiltration relationships, leads to greater infiltration in the interspaces, breakdown in patch condition and increases the time period required for the restoration of patches. Removal of the crust is also likely to affect ecosystem processes such as plant germination, establishment, survival and nutrient mineralization (Eldridge, 2001). The combination of reduced crust cover and low density of patches in both active (ploughed) treatments likely compromises the ability of these landscapes to capture and store essential resources such as water and nutrients, prolonging the recovery process.

4.2. Treatment effects on understorey plants and shrubs

Grazing exclusion in grasslands typically leads to a recovery of perennial grass cover (Tiedemann and Klemmedson, 2004; Mata-González et al., 2007), but the time over which this occurs varies widely and is related to the capacity of the landscape to respond to rainfall. Consequently, many studies have failed to detect increased perennial grass density after more than a decade of removal of livestock (West et al., 1984; Valone et al., 2002) or even after physical removal of shrubs and exclosure (active conservation; Roundy and Jordan, 1988). In our study, increased disturbance was associated with significant declines in plant diversity and changes in composition, consistent with our second prediction. Plant cover and diversity were greatest under active conservation (disturbance followed by exclosure) in contrast to grazing post-disturbance (Jauffret and Lavorel, 2003). However, the substantial recovery of perennial grasses at the study site only two years after treatment (Robson, 1995) was not observed in our study 18 years after treatment and can only be attributed to reduced summer rainfall since ploughing. This suggests to us that the greater cover of grazing-sensitive perennial bunch grasses such as *E. eriopoda* and *Monachather paradoxa* that we recorded in the ungrazed-unploughed treatments was due mainly to removal of grazing rather than to summer rainfall *per se*. Rainfall records for the 18 year period since treatment confirms our suggestion that rainfall was less effective than grazing (Electronic Appendix Fig. 1). Differences in post-treatment rainfall therefore likely account for the wide range of responses to woody shrub removal by ploughing in eastern Australia. This reinforces the importance of managing stocking rates after shrub removal, particularly in areas where the soils are highly erodible and natural rates of surface stability are inherently low.

4.3. Treatment effects on shrubs

Studies of shrub density and richness worldwide indicate a range of responses to both the individual and combined effects of grazing and physical shrub removal. In our study shrub densities were two and a half – to three-times greater under the active pastoral treatment (ploughing and grazing, 5750 shrubs ha^{-1}) than either the conventional pastoral (grazing only, 2300 shrubs ha^{-1}) or active conservation (3400 shrubs ha^{-1}) treatments. The density of live shrubs on ploughed-grazed plots was two and a half time greater than that on grazed plots that had not been ploughed, even if we take into account shrubs that had died over the past 18 years. There were some highly variable results, particularly in the active pastoral treatment, which could have been due to differences in the initial species composition or soil heterogeneity at the landscape scale. One of the active pastoral plots supported a large number of *E. gilesii* seedlings which were not found at the other two plots. *E. gilesii* are profuse seed producers, unpalatable to stock and able to regenerate by root suckering (Burrows, 1974). It is not surprising that a large number of seedlings can be found in the area that was ploughed and grazed and previously dominated by this species. Our data therefore suggest a trend of greater density of shrubs after ploughing, particularly in the area dominated by species that are able to regenerate by root suckering or epicormic buds. The results suggest a trend of greater density of shrubs after ploughing, even though this difference was not statistically significant because of limited replication at the landscape scale (three landscape-level replicate exclosures). We detected a trend of declining landscape function indices, and probably irreversible structural changes over a period of declining summer rainfall. Given the likely reductions in soil nutrients and depletion of soil seed banks of perennial grasses (Hodgkinson and Harrington, 1985), our study suggests that shrub-encroached woodlands are unlikely to revert to grassland in the next century (e.g., Jeltsch et al., 1997). Indeed our observations are consistent with widespread reports in the literature of failure of mechanical treatments to provide long-term shrub control. Instead of recovering to grasslands, shrubs continued their domination and left the remaining degraded area bare (Roundy and Jordan, 1988; Ruthven III et al., 1993; Mata-González et al., 2007).

As predicted, the effects of ploughing and grazing on shrub density were highly species-specific, with pastoral (grazed) plots supporting fewer *A. aneura*, *Acacia burkitii* and *D. viscosa* subsp. *angustissima* shrubs that are moderately palatable to livestock and macropods (kangaroos), particularly when young (Hodgkinson, 1979; Tiver and Andrew, 1997). There were also fewer *E. longifolia* shrubs under the plough-graze scenario, but not under the plough-ungraze scenario. This result was not surprising given that *E. longifolia* readily reproduces by disturbance-induced root suckering (Chinnock, 2007). *E. longifolia* is also highly palatable to livestock, thus resprouting shrubs would be expected to have been not only more accessible to browsers, but also to have contained higher leaf nitrogen or phosphorus or lower concentrations of grazing suppressing polyterpenes and flavones (Chinnock, 2007), which would discourage vertebrate browsers. Higher concentrations of plant defence substances are known to be present in regrowth leaves in response to heavy browsing (e.g., *Acacia berlandieri*; Forbes et al., 1995).

Ploughing failed to reduce the density of *Dodonaea viscosa,* one of the most abundant shrubs in eastern Australia, and a species regarded as being moderately susceptible to mechanical control (Harland, 1993; Robson, 1995; Eldridge and Robson, 1997). Ploughing did, however, reduce *E. sturtii* densities almost eightfold, with surviving shrubs restricted almost entirely to the two unploughed treatments, and with few shrubs greater than 1.5 m tall. Ploughing would be expected to stimulate regeneration of *E. sturtii* through root suckering or from epicormic buds close to the soil surface (Hodgkinson, 1979; Chinnock, 2007). As subsequent grazing would also favour its regeneration by removing competitors, *E. sturtii* would likely dominate areas with a history of heavy stock grazing (Tiver and Andrew, 1997), unlike its congener *E. longifolia*. Ploughing provides effective control of *E. sturtii* only where the depth of ploughing is within the range of 20–40 cm (Harland, 1992). The different responses of *E. sturtii* and *E. longifolia* to ploughing in our study therefore could have resulted from differences in the depth of ploughing as a result of the uneven micro relief of shrub patches. Ploughing at shallow depths, by failing to cover remaining roots with soil, is known to elicit bud formation and resprouting in some shrubs (Wiedemann and Kelly, 2001). Lower levels of re-establishment of *E. sturtii* compared with *E. longifolia* in the ploughed-ungrazed plots could also have been caused by the association of *E. longifolia* with arbuscular mycorrhizal fungi, which is not found in *E. sturtii* (O'Connor et al., 2001).

The distribution of shrub sizes varied substantially in relation to grazing, reinforcing our view that, rather than providing long-term control of shrubs, grazing of ploughed shrubland is likely to skew the distribution of shrubs to one dominated by a larger number of smaller shrubs, in our case, large cohorts of *E. gilesii* seedlings. *E. gilesii* is a prolific seed producer and typically forms dense monospecific stands of up to 12,000 shrubs ha^{-1} (Burrows, 1974). In the short-term, a larger number of smaller shrubs may be a positive management outcome, as smaller shrubs will likely make access and stock movement easier, and would not suppress understorey plants to the same extent as large mature shrubs with extensive canopies.

4.4. Management of shrub-encroached woodlands

The results of this study question the long-term viability of shrub removal using blade ploughing on sandy soils in eastern Australia, and therefore the benefits of investing in mechanical shrub control. There were few clear pastoral benefits of shrub treatment 18 years after treatment, though we acknowledge that some grazing value, in the form of enhanced productivity of pastorally important perennial grasses, may have accrued at various times over the past 18 years (e.g., Robson, 1995). The anticipated recovery of perennial grasses with shrub removal did not occur, and the reasons for this are likely numerous such as prolonged grazing from sheep and feral goats, and below average periods of summer rainfall. Our results indicate that ploughing was not an effective method to control shrubs in this area given that the results are species-specific and likely to depend on post-treatment rainfall (Robson, 1995). In combination with grazing, ploughing promoted a groundstorey plant community dominated by short-lived, grazing-tolerant species at the expense of perennial grasses. Furthermore, shrub removal by ploughing had generally negative effects on the soil, reduced patches and cryptogamic cover even under exclosure. More importantly, ploughing failed to sustain the increases in grass cover, and resulted in long-term increases rather than reductions in shrub density. In the context of pastoralism, alternative strategies such as adjusting the livestock number, or the use of shrublands for short-term agistment of livestock during productive (La Niña) seasons, a contemporary form of transhumance, or the use of alternative grazing animals such as goats may be more appropriate. An alternative perspective, which is gaining traction worldwide, is to value these encroached shrublands in terms of the ecosystem benefits that they can provide for communities such as fresh water, habitat, fuel wood and sinks for carbon dioxide. We acknowledge, however, that such a shift will likely adversely impact the livelihoods of producers at a time when they are attempting to the recoup the costs of these new strategies.

5. Conclusion

We found that that ploughing was generally an inappropriate method for reducing shrub densities in encroached Australian semi-woodlands. The effects of ploughing (with and without grazing) were largely species-specific. Given that Australian semi-arid shrublands are rarely monospecific, broad-scale indiscriminate treatments such as ploughing will provide effective control of only a limited number of species, and could conceivably select for shrubs that resprout after ploughing. The results of our study provide valuable insights into the problems associated with mechanical shrub control in eastern Australia. More studies are needed to assess the efficacy of shrub control with mechanical methods, particularly given the considerable amount of money spent on 'managing' shrub-encroached woodlands over the last two decades.

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Appendix. Supplementary data

The supplementary data associated with this article can be found in the on-line version at doi:10.1016/j.clnu.2010.06.001.

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