Factors influencing early restoration progress of a *Eucalyptus tereticornis* open forest on former agricultural land

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We monitored an area that was revegetated with the goal of restoring a *Eucalyptus tereticornis* open forest on former agricultural land in central, eastern Queensland. Revegetation involved: (1) planting 60 ha of previously cleared and heavily grazed land with eight local trees species; and (2) removing cattle grazing to encourage natural regeneration in areas where some mature trees remained. We compared the revegetation site to native pasture that had also been previously cleared, with only scattered paddock trees remaining, and continued to be managed for livestock production (an area similar to the revegetation site, prior to planting) and a remnant forest (reference area). Nine years since revegetation began there was some evidence that the revegetated site was diverging from pasture in terms of understorey plant composition, sapling density and topsoil C and N. There was little divergence in terms of plant species richness (native, introduced, grass, forb and woody plant richness), herbaceous biomass and woody plant regeneration. Some monitoring plots were subject to fire (prescribed fire and or wildfire) over the period of monitoring. With increasing time since fire, the richness of native species, introduced species and grass species (both native and introduced) declined, and forb and grass species richness declined with increasing litter biomass, suggesting that the occurrence of fire and the associated removal of litter biomass has a positive influence on herbaceous diversity in this ecosystem. Woody plant regeneration persisted through lignotubers at the revegetation site and at the pasture, but this regeneration was stunted at the pasture presumably due to livestock grazing. Hence areas of former *E. tereticornis* forest showed promising regenerative capacity where mature trees remained and where livestock grazing was removed.

Key words: species richness, plant composition, native tree plantings, soil carbon, vegetation offsets, regeneration, fire, grazing.

INTRODUCTION

REVEGETATION is a key strategy in the conservation of endangered grassy woodlands (Lindenmayer *et al.* 2010). In Australia, open forests and woodlands with a grassy understorey often occur in productive areas of the landscape and have been heavily cleared or modified for agricultural production. Open *Eucalyptus tereticornis* forests and woodlands have been extensively cleared from eastern Queensland, New South Wales and northern Victoria and those on alluvial soils are currently endangered in south-east Queensland (Vegetation Management Regulation 2000). Hence, mature intact *E. tereticornis* stands are particularly rare in the landscape but are important because they provide habitat for a range of ecologically significant fauna, largely through hollow-forming trees and floral resources (White 1999; Dobson *et al.* 2005; Smith *et al.* 2007). To ensure the conservation of *E. tereticornis* ecosystems and their associated fauna some revegetation efforts will undoubtedly be important in the future. Currently, however, published studies on the ecology of *E. tereticornis* ecosystems and on their regenerative potential are scarce.

The success of revegetation efforts is fundamental to the viability of vegetation offset schemes, where vegetation loss is offset by gains through the restoration of similar vegetation elsewhere. The height of the Awoonga dam, an important water storage facility in central, eastern Queensland (Fig. 1), was raised in June 2002 to increase storage capacity. Consequently, the clearing and loss of approximately 130 ha of native forest dominated by *E. tereticornis* occurred as a result of the dam expansion. To offset this loss a project was established to revegetate former agricultural land with flora typical of *E. tereticornis* forests, and this presented an opportunity to establish a long-term monitoring programme. Long-term monitoring of revegetation success is important given that offset agreements are often criticized due to time lags between vegetation loss and gain, inadequate compliance and lack of evidence of net gain in vegetation to compensate for vegetation loss (Gibbons and Lindenmayer 2007). Equally as important, the functional equivalence of the vegetation being protected, enhanced or restored is not always comparable to the vegetation lost (Race and Fonseca 1996; Gibbons and Lindenmayer 2007; Burgin 2008), resulting in a net loss of habitat.

Given the potentially large number of plant species occurring in remnant *E. tereticornis* forest (e.g., Bean *et al.* 1998) and the constraints associated with restoration of an entire plant

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community on degraded agricultural land, revegetation aimed to develop a structural framework of overstorey trees with the goal of restoring natural recovery processes. As tree canopy cover exerts an influence on understorey vegetation composition (Gibbs *et al*. 1999; Butler *et al.* 2006; Cummings and Reid 2008) and can aid in the dispersal of plant species (Keenan *et al.* 1997; Reay and Norton 1999; Selwyn and Ganesan 2009) we predict that plant community assemblages typical of natural *E. tereticornis* forest will develop over time. To encourage understorey development livestock grazing was removed from the revegetation site, as grazing is known to have a negative impact on regeneration in many cases (e.g., Spooner *et al*. 2002; Briggs *et al*. 2008; Griscom *et al.* 2009). Occurrence of fire (wildfire and prescribed fire) at the revegetation site has a potentially confounding influence on native plant regeneration and composition (e.g., Whelan 1995) but fire could provide a useful management tool in encouraging natural regeneration (Pyke *et al.* 2010). In fact, remnant

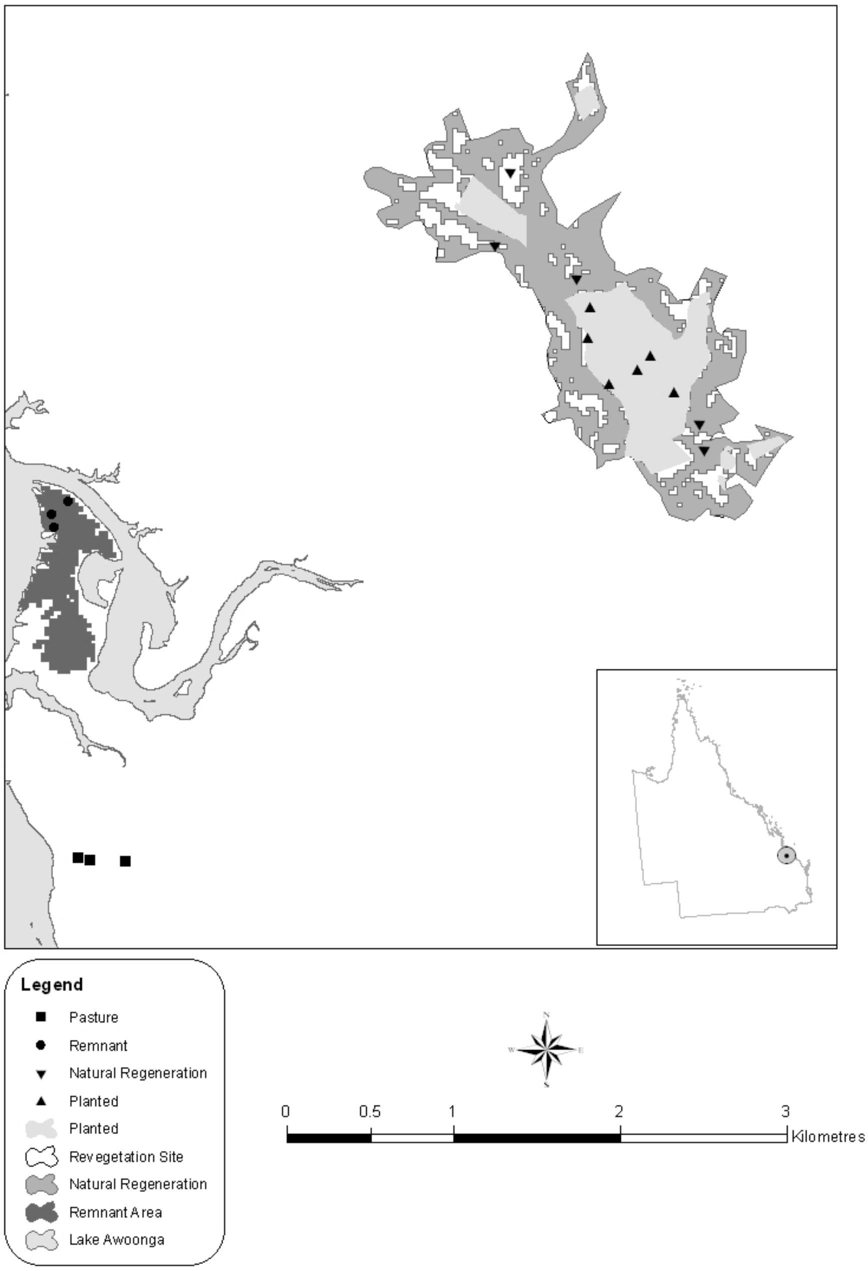

Fig. 1. Location of the revegetation site relative to Lake Awoonga, and locations of the monitoring plots in the different treatments (planted, natural regeneration, remnant and pasture). Forest cover (shaded areas) at the remnant area and natural regeneration area (within the revegetation site) is mapped based on Statewide Land and Tree Study data for 2005.

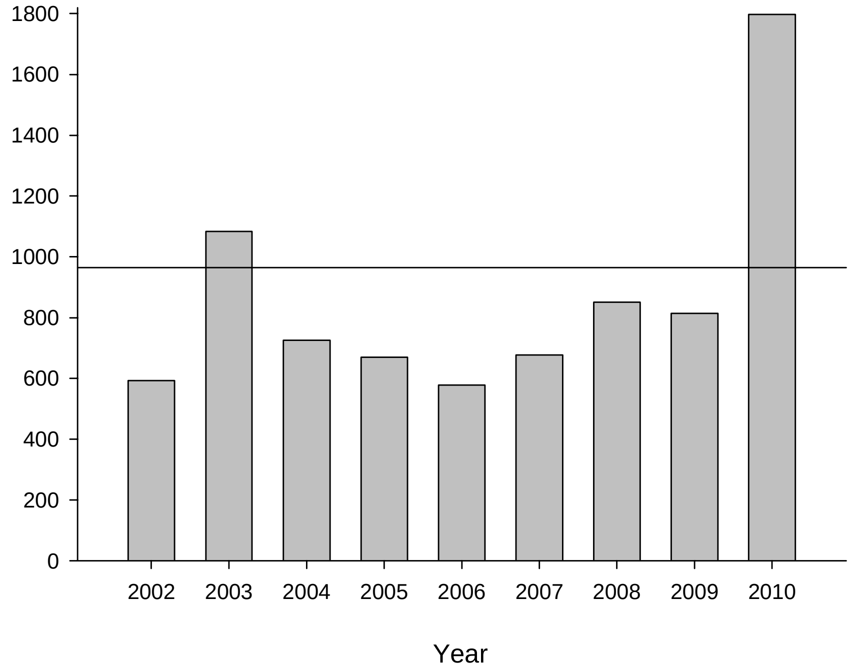

Fig. 2. Total yearly rainfall 2002–2010 for the revegetation site relative to the mean annual rainfall (1889–2010), represented by the solid horizontal line.

E. tereticornis forest is thought to be resilient to low intensity fire, and burning intervals of 3–6 years are currently recommended to encourage biodiversity in this ecosystem (Environment, Climate and Resource Sciences 2012).

Vegetation development is likely to be influenced by both biotic and abiotic factors; hence, potential alterations to topsoil fertility due to land use history may represent a barrier to revegetation progress (Prober *et al.* 2002b; Cummings *et al.* 2005). For example, soil C and N is initially predicted to decrease following afforestation (O'Connell *et al.* 2003; Paul *et al.* 2003; Kirschbaum *et al.* 2008) and this could have an impact on understorey vegetation composition (Smallbone *et al.* 2007). A review by Paul *et al.* (2002) suggested that recovery of soil C may take 30 years following afforestation, but there was considerable variation depending on forest type, making it difficult to predict soil C changes across ecosystems. Hence, the time frames for litter development, decomposition and the associated changes in topsoil C and N are largely unknown for *E. tereticornis* forest but need to be better understood to assist revegetation efforts of this ecosystem.

In this paper we report results approximately nine years after revegetation was initiated. To determine whether the revegetation site was following the desired restoration trajectory we compared the revegetation site (planted areas and areas with natural regeneration) to a pasture (previously *E. tereticornis* forest cleared for grazing) and a remnant *E. tereticornis* forest, over time. As restoring former agricultural land to native forest is a challenging proposition that may take several decades, if not longer to carry out (Reay and Norton 1999; Vallauri *et al*. 2002; Wilkins *et al*. 2003; Standish *et al*. 2007) it is too early to expect similarity between the revegetation site and the remnant forest. However, based on the above-mentioned literature we predicted that changes in the revegetation site would occur over time as tree canopy cover increased, and that some changes would be apparent within the first nine years. Hence we aimed to determine if and how the revegetation site on former agricultural land was diverging from the cleared pasture through time. The effects of disturbance agents, such as livestock grazing and fire on woody plant regeneration and understorey composition are discussed.

MATERIALS AND METHODS

Study area description

The revegetation site is a former farming property approximately 2 km from Lake Awoonga (Fig. 1). The main historical land use for the site was livestock production, which began in the early 20th century and resulted in the plains and low hills being largely cleared. Scattered large *E. tereticornis* trees that are present throughout the cleared area suggest that the vegetation of the site was open forest or woodland dominated by *E. tereticornis* prior to clearing. Remnant *E. tereticornis* open forests and woodlands on alluvial plains may contain several other tree species, such as *Eucalyptus crebra, E. moluccana, E. melanophloia, Angophora* species, *Lophostemon suaveolens, Corymbia tessellaris* and *C. intermedia*, and often have a grassy understorey containing a range of native grasses and forbs (Bean *et al*. 1998; Young and Dillewaard 1999; T. Lewis, unpublished data). Lignotuberous tree regeneration is also common in the understorey of these remnants (e.g., up to 1 000 stems per hectare). Soils at the revegetation site are largely formed from local alluvium and are strongly duplex with a silty clay loam surface overlying a mottled yellow to brown medium clay subsoil. The site comprised of native pasture that had been heavily grazed prior to revegetation (Appendix 1a). An initial vegetation survey of the planted area in 2003 (unpublished data) recorded high proportions of bare ground (26–33%), and a pasture dominated by *Cynodon dactylon*, *Chloris truncata*, *Eragrostis brownii* and *Panicum effusum*.

Average annual rainfall for the site is 965 mm (1889–2010) based on spatially interpolated Bureau of Meteorology data (Jeffrey *et al.* 2001). Rainfall is summer dominant with highest monthly averages in January and February (174 mm in both cases) and the lowest monthly averages in August and September (29 mm and 34 mm, respectively). Rainfall has been below average for all years since 2002 with the exception of 2003 and 2010 (Fig. 2). Average day time temperatures vary from 31ºC in December and January to 22ºC in July and average night time temperatures vary from 21–22ºC (December to February) to 10ºC in July. Several frosts can occur during some winters.

Revegetation methodology

Planting occurred over 60 ha on previously cleared land in 2002 and 2003 (30 ha each year). A mix of native tree species were planted, reflecting tree species composition in nearby remnants. Tree species planted (and their proportional representation) were: *Eucalyptus tereticornis* (57%), *E. moluccana* (6%), *E. crebra* (2%), *E. melanophloia* (1%), *Corymbia tesselaris* (16%), *C. intermedia* (2%), *Acacia disparrima* (13%) and *Lophostemon sauveolens* (3%). Seed was collected from local provenances between September and March and local provenance seed was purchased where seed was not available. Seeds were stored in a cold room at 4ºC prior to planting. Seedlings were propagated in a nursery near Gympie; seed was sown into 200 mL pots in a potting mix of 50% pine bark, 25% peat and 25% sand and 2.5 kg/m^3 of Osmocote. Seedlings were grown for approximately four months prior to planting, initially in a glasshouse and then under full sunlight in preparation for field conditions.

Site preparation was carried out prior to planting; planting rows were wing ripped to a depth of 50 cm and offset disc ploughed to create 2 m wide planting rows. A modified contour pattern was followed to minimize soil erosion, assist with moisture retention and provide a more natural planting configuration. Herbicide was applied before (glyphosate, 450 g/L at 4 L/ha) and after planting (simazine 500 g/L at 6 L/ha) to reduce competition from herbaceous vegetation and fertilizer (Incitec Starterphos) was applied immediately following planting at a rate of approximately 100 g per tree. Trees were planted at variable spacing to mimic natural variability in regeneration. Densities ranged from 310–680 trees per hectare (average 430 trees per hectare). A deliberately higher density of trees was planted than that of the nearby remnant *E. tereticornis* forest (density of tress with a DBH >10 cm in the remnant area was 97 trees per hectare), as some mortality was expected in the establishment phase. However, survival of planted trees was high (>90%) and hence thinning of the planted trees took place between April and June 2009 over 65% the planted area. Trees were cut at their base with a chainsaw and glyphosate was applied to the cut stump (450 g/L, 1 part glyphosate to 20 parts water). Follow-up herbicide was used on resprouting foliage, where necessary. An average of 85 trees per hectare was removed through thinning and the pre-thinning proportions of species and size classes were retained after thinning. All thinned stems were left on site. While natural self-thinning would occur, such a process is slow, and hence thinning treatments were applied to hasten this ecological process and encourage development of desirable habitat attributes (as recommended by Vesk *et al.* 2008).

In this paper 'the revegetation site' includes both planted areas and areas where natural regeneration of the woody understorey has been encouraged. No planting was carried out where clearing had been less intensive and several mature *E. tereticornis* trees remained in the overstorey. In these areas natural regeneration was encouraged by removal of livestock grazing. Livestock grazing was excluded from the entire revegetation site in 2001, but a small number of cattle (<40 head) have been observed at the site on several occasions since. Thus, the established monitoring plots were fenced in 2005 to eliminate potential effects of livestock grazing on revegetation progress. To aid native vegetation recovery, ongoing control of declared weeds using herbicides has taken place at the revegetation site since 2002.

Monitoring methodology

Monitoring was carried out between 2005 and 2011 on a yearly basis. A total of 17 monitoring plots were established within four different treatment areas (Fig. 1). Six plots were located in the planted area (3 plots in each of the 2002 and 2003 plantings), five plots in the natural regeneration area, three plots in a nearby native pasture (representative of the site prior to any revegetation) and three plots in a nearby remnant *E. tereticornis* forest (Fig. 1). These treatment areas were selected using vegetation mapping and on-ground assessments. The remnant treatment was mapped remnant *E. tereticornis* forest (i.e., where the dominant canopy had >70% of the height and >50% of the cover relative to the undisturbed height and cover of that stratum and was dominated by species characteristic of the vegetation's undisturbed canopy; Accad *et al*. 2008) that was >5 ha in area and was nearby the area that was cleared when the dam wall was raised. The natural regeneration treatment differed to the remnant treatment in terms of large tree density and history of livestock grazing. The natural regeneration area contained <5 trees per hectare with DBH >50 cm, while the remnant area contained =30 trees per hectare with DBH >50 cm. The natural regeneration area had a history of more intense livestock grazing than the remnant, prior to removal of livestock from the revegetation site. The pasture area was selected as an area representative of the revegetation site prior to revegetation (i.e., previously cleared *E. tereticornis* open forest) as no pre-treatment data was collected at the revegetation site. The pasture area was a native pasture that had scattered *E. tereticornis* paddock trees and was grazed by cattle. The remnant and pasture treatments were selected based on the closest available sites to the revegetation site that were considered comparable in terms of their position in the landscape and soil type.

Fire was used in management of the property prior to revegetation, with areas surrounding the cleared plains being burnt in spring every three years on average. Several fires occurred at the monitoring plots (wildfire and prescribed fire) since 2001. All natural regeneration plots were burnt by unplanned fires in January 2003, two natural regeneration plots were burnt in June 2005, and in January 2007, two natural regeneration plots that were not burnt in 2005, and all three of the remnant plots were burnt. The intensities of these fires were not measured, but observations and details of the weather conditions during the fires suggest they were of low to moderate intensity (<2 500 kW/m). Prescribed fire was used in the planted area in September 2008, which burnt all planted plots. This was a low intensity fire, conducted in the evening to minimize scorch on the planted trees.

Monitoring plots were circular and 0.1 ha in area. All monitoring plots were fenced to exclude cattle grazing with the exception of the pasture plots. The pasture area was leased by a cattle grazier. This treatment was periodically grazed between 2005 and 2011 at stocking rate of approximately 0.2 cattle per hectare.

Tree growth and regeneration

Measurements of tree growth and regeneration were carried out at monitoring plots in September–November from 2005 to 2010. All individual trees and shrubs >1 m in height were tagged with a unique identifier. All trees and shrubs 0.5–1 m in height were also tagged in 2005 to follow this regeneration cohort through time. A count of all trees and shrubs of 0.5–1 m in height was recorded for each species but new recruits after 2005 were not tagged. We defined woody plant regeneration as all trees and shrubs with a DBH of <5 cm and a height >0.5 m, while saplings were defined as trees with a DBH =5 cm but <10 cm. The total height, diameter (DBH, measured at 1.3 m) and species was recorded for each tagged individual during each measure.

Understorey vegetation

Floristic composition and richness was assessed in March–April from 2005 to 2011. Vegetation was assessed within four 1 m^2 quadrats in each plot. Each quadrat was positioned approximately 8 m from the centre of the plot along major compass bearings (N, S, E and W). Quadrats in the planted plots were positioned between planting rows to minimize the effects associated with row disturbance (i.e., ploughing and herbicide application). All vascular plant species growing in each quadrat were recorded and assigned a modified Braun-Blanquet cover abundance score ranging from 1 (<5% cover, with isolated occurrences) to 10 (100% cover). Plant species richness was calculated for each plot (i.e., richness over four quadrats). Nomenclature follows Bostock and Holland (2007) and recent taxonomic revisions accepted by the Queensland Herbarium. Introduced species are indicated with an asterisk throughout the text.

Soil and ground cover biomass

Soil samples were collected from each treatment area in March–April (2005–2011). Twenty samples per plot were randomly collected with a 2-cm diameter push corer to 10 cm depth and were bulked for analysis. Samples were air-dried and ground to <2 mm prior to analysis. We report results from analysis of total percentage C (Heanes method) and N (Kjeldahl digestion) in the topsoil, as these variables are associated with nutrient cycling. Details of the analytical methods are provided in Collins (2000).

Grass and litter biomass was estimated for each plot using a modified version of the ranked set technique of McIntyre (1952). This involved collecting samples where biomass was visually ranked as low, medium and high at each plot and calculating an average of these samples. Living plant material and dead plant material were collected separately at each sample point from within a 0.5 × 0.5 m quadrat. All samples were dried in an oven at 70–80°C to a constant weight and weighed.

Environmental variables

Nine predictors of plant species richness and composition, soil C and N, woody plant regeneration or ground cover biomass were selected: clay and fine sand percentages in the topsoil, topsoil C:N, time since fire, tree basal area, grass biomass and litter biomass, rainfall in the six months prior to sampling and forest cover surrounding each plot (Table 1). Other variables were investigated (e.g., percentage bare ground, % coarse sand and silt in the topsoil) but due to correlations with the above variables, these were excluded from our analysis.

Analysis

Multivariate analyses were carried out for plant cover (average cover score for each species per plot) across all sampling times. We tested whether species composition on former agricultural land was diverging from the pasture area and converging towards remnant *E. tereticornis* forest through time using CANOCO, version 4.5 (ter Braak and Šmilauer 2002).

Preliminary detrended correspondence analysis confirmed that species responses to environmental predictors were more linear than Gaussian. Accordingly, direct gradient analysis (redundancy analysis, RDA) was used to explain the plant abundance data using selected explanatory variables. Treatment areas (natural regeneration, planted, remnant and pasture), time (Julian time) and environmental variables (Table 1) were defined as explanatory variables in the analysis. While landscape-scale effects are undoubtedly important in determining the distribution of mature *E. tereticornis* forest, all but one predictor was at the site-scale given the relative importance of site-based variables in determining changes in woody plant regeneration and composition (Debuse *et al.* 2009). Monte Carlo permutation tests were used to test the significance of each variable. Variables that had no significant influence on plant composition ($P < 0.05$) were dropped from the analysis. Partitioning of variance was used to determine the variance described by treatment areas and other environmental variables. This involved the use of partial ordination by accounting for confounding of effects of treatment area and other environmental

Table 1. Definitions of the variables used in analysis (treatment and covariates) and the variation (minimum and maximum values) across all plots and times. Covariates used in each univariate analysis are provided in the definition.

Variable	Definition
Treatment area: Pasture Planted Natural regeneration Remnant	Treatment areas were selected using vegetation mapping and on-ground assessments. Refer to text for more information on how these areas were defined.
% fine sand in topsoil	% of particles 0.02–0.2 mm in the 0–10 cm soil layer (min = 9.5, max = 25.9). Covariate in analysis of herbaceous biomass, litter biomass, topsoil % C and % N.
% clay in topsoil	% of particles <0.002 mm in the 0–10 cm soil layer (min = 20.0, max = 55.9). Covariate in all analyses.
topsoil C:N	Ratio of total % carbon to nitrogen in the 0–10 cm soil layer (min = 11.8, max = 17.7). Covariate in all analyses, except for topsoil % C and % N.
Time since fire	Number of years since the last fire (wildfire or prescribed burn) (min = 0.5, max = 16.0). Covariate in all analyses.
Tree basal area (m^2/ha)	The cross-sectional area that trees occupy at 1.3 m above ground, based on measurement of DBH of all trees >2 cm diameter in 0.1 ha monitoring plots (min = 0.0, max = 32.4). Covariate in all analyses.
Herbaceous biomass (t/ha)	Oven dried weight of herbaceous vegetation <1 cm in diameter collected from 0.5 × 0.5 m quadrats (min = 0.2, max = 13.1). Covariate in all analyses, except for herbaceous biomass and litter biomass.
Litter biomass (t/ha)	Oven dried weight of leaf litter <1 cm in diameter collected from 0.5 × 0.5 m quadrats (min = 0.1, max = 13.8). Covariate in all analyses, except for herbaceous biomass and litter biomass.
Recent rainfall (mm)	Rainfall in the 6 month period prior to sampling, based on interpolated Bureau of Meteorology data (min = 276, max = 1354). Covariate in analysis of topsoil % C and % N. This response variable was explained by the variation in time in our analyses.
Surrounding forest cover (%)	Forest cover in a 500 m radius around each plot, based on Statewide Land and Tree Study data for 2005 (Kuhnell *et al.* 1998) (min = 11.5, max = 96.7). Covariate in all analyses, except for herbaceous biomass, litter biomass, topsoil % C and % N.

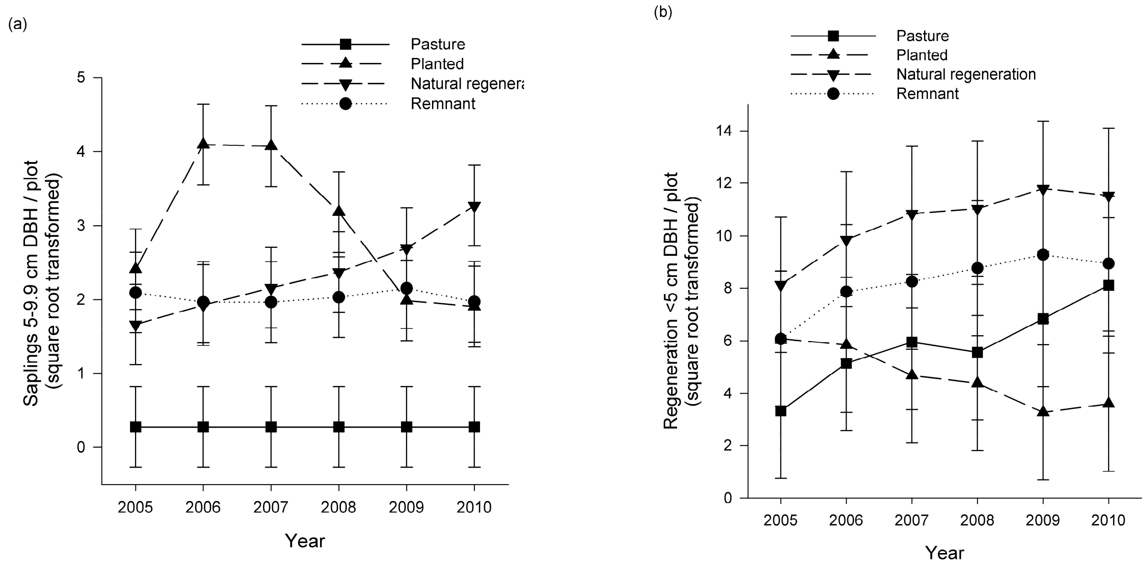

Fig. 3. The density of: (a) saplings with a DBH =5 cm but <10 cm per 0.1 ha plot and (b) regenerating woody plants with a DBH of <5 cm and a height >0.5 m per 0.1 ha plot, in the different treatment areas through time. Data points are predicted square root transformed means adjusted for covariates (± SE of differences of means, based on comparisons between the planted plots, n = 6 and the pasture and remnant plots, n = 3). Note: thinning of saplings took place between the 2008 and 2009 measures in the planted plots.

variables, through the use of covariates (Borcard *et al.* 1992). Three RDA runs were carried out to determine: (1) pure effect of treatment area; (2), pure effect of environmental variables (including time); and (3) combined variation due to confounding effects of treatment area and environmental variables. The remaining variation was unexplained. Separate RDA runs were also run for the 2005 and 2011 data sets to assess changes in plant cover between the first measure and the most recent measure. To test for significance in the variation in community composition among treatment areas in both 2005 and 2011 we used one-way ANOSIM (analysis of similarity) in PRIMER (Clarke and Gorley 2001). The data matrix consisted of pairwise comparisons among plots based on the Bray-Curtis similarity index. The R statistic generated by ANOSIM provides a relative measure of separation of the treatment areas.

Univariate analyses were carried out on data collected from 2005–2011 in GenStat (11th edition) to determine differences between treatment areas (planted, natural regeneration, remnant and pasture) through time and the influence of covariates (Table 1) on 13 different ecosystem measures. These were: native and introduced plant species richness, richness of plant groups (forbs, grasses and woody plants), abundance of two common grasses (*Cynodon dactylon* and *Heteropogon contortus*), total percentage C and N in the topsoil, biomass of herbaceous vegetation, leaf litter biomass, the density of woody plant regeneration and sapling density. We used general analysis of variance (ANOVA), for which the treatment structure was Treatment × Sampling time and the block structure was defined as Plot/Sampling time. Covariates used in each analysis varied depending on the response variable of interest. Only relevant covariates that were likely to impact on each response variable were used (Table 1). Covariates that varied spatially but did not vary temporally were assigned to the plot stratum, while covariates that varied both spatially and temporally (topsoil C:N, tree basal area, herbaceous and litter biomass and time since fire) were included in both the plot and plot.time strata. Only covariates with $P < 0.1$ were included in the final analysis. In our analyses the variation explained by rainfall in the six months prior to sampling was explained by the variation in time, such that no variation remained to explain rainfall in the plot.time strata. Standardized beta coefficients were calculated to compare the importance of the predictor variables. Plant species richness and abundance data, woody plant regeneration and sapling data and herbaceous and litter biomass data were square-root transformed to satisfy the assumptions of ANOVA.

RESULTS

Woody plant regeneration and structure

Mean height in November 2010 across all planted species was 8.4 ± 0.32 m, and tree growth increased linearly through time. Sapling density varied significantly among treatments ($F_{3,12}$ = 14.28, P <0.001) with lower densities of saplings recorded in the pasture plots than in the other treatments (Fig. 3a). However, the treatment effect varied significantly over time ($F_{15,64}$ = 6.32, P <0.001). There was an initial

Table 2. Repeated measures ANOVA results for the effects of treatment (pasture, planted, natural regeneration and remnant), sampling time and the treatment area by time interaction and covariates (% clay in the topsoil, topsoil C:N, tree basal area, forest cover, herbaceous biomass, litter biomass and time since fire) on the richness of native, introduced, forb, grass and tree and shrub species, and the abundance of *Cynodon dactylon* and *Heteropogon contortus* between 2005 and 2011. Only covariates with P <0.1 were included in the final analysis. *P <0.05, **P <0.01, ***P <0.001.

	Species origin		Species life form			Abundance of two common grasses	
Source of variation	**Native**	**Introduced**	**Forb**	**Grass**	**Tree and shrub**	***Cynodon dactylon***	***Heteropogon contortus***
Plot stratum							
Treatment	$F_{3,7}$ = 14.49**	*ns*	$F_{3,11}$ = 3.95*	$F_{3,10}$ = 14.17***	$F_{3,13}$ = 13.47***	$F_{3,8}$ = 17.25***	*ns*
Covariates	*ns*	*ns*	*ns*	$F_{3,10}$ = 3.45	*ns*	*ns*	$F_{4,9}$ = 3.79*
% clay	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*
C:N	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*
Tree basal area	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*
Forest cover	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*
Herbaceous biomass	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*	$F_{1,9}$ = 9.54*, β = 1.78
Litter biomass	*ns*	*ns*	*ns*	$F_{1,10}$ = 5.37*, β = –2.29	*ns*	*ns*	*ns*
Time since fire	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*
Plot.Time stratum							
Time	$F_{6,63}$ = 9.56***	$F_{6,75}$ = 6.64***	$F_{6,74}$ = 8.22***	*ns*	*ns*	$F_{6,64}$ = 3.03*	$F_{6,72}$ = 14.07***
Time×Treatment	$F_{18,63}$ = 2.23*	$F_{18,75}$ = 2.72**	$F_{18,74}$ = 2.70**	$F_{18,73}$ = 3.84***	*ns*	$F_{18,64}$ = 5.93***	$F_{18,72}$ = 15.59***
Covariates	$F_{3,63}$ = 3.92*	$F_{1,75}$ = 9.90**	$F_{2,74}$ = 3.20*	$F_{3,73}$ = 14.20***	*ns*	$F_{2,64}$ = 5.84**	$F_{4,72}$ = 4.39**
C:N	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*
Tree basal area	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*	$F_{1,72}$ = 8.67**, β = –3.13
Herbaceous biomass	*ns*	*ns*	*ns*	*ns*	*ns*	*ns*	$F_{1,72}$ = 5.55*, β = 2.79
Litter biomass	*ns*	*ns*	$F_{1,74}$ = 6.05*, β = –2.46	*ns*	*ns*	*ns*	*ns*
Time since fire	$F_{1,63}$ = 7.13*, β = –2.65	$F_{1,75}$ = 9.90**, β = –3.14	*ns*	$F_{1,73}$ = 38.80***, β = –5.66	*ns*	$F_{1,64}$ = 8.46**, β = 2.91	*ns*

increase in sapling density in the planted area, as there was transference of planted trees from regeneration into saplings, which was not apparent in the other treatment areas and an increase in sapling density in the natural regeneration area over time that was not apparent in the other treatment areas (Fig. 3a). Of the covariates, only tree basal area had significant influence on sapling density across all years combined ($F_{1,12}$ = 10.41, P = 0.007); greater basal areas were associated with lower sapling densities (β = –3.20).

Woody plant regeneration varied from 1–414 plants per 0.1 ha, across all treatment areas and years. There was a significant treatment area by time interaction for regeneration density ($F_{15,65}$ = 4.06, P <0.001). This was due to a decrease in regeneration density in the planted area over time as planted regeneration grew and moved into the sapling cohort and an increase in regeneration density over time in the other treatment areas (Fig. 3b). There was a trend of higher densities of regeneration in the natural regeneration plots than in the pasture and planted plots across all years ($F_{3,13}$ = 2.98, P = 0.070; Fig. 3b). None of the covariates measured had a significant influence on regeneration density.

Understorey vegetation composition and richness

We recorded a total of 161 plant taxa across the four treatment areas. Native species richness of the understorey vegetation ranged from 9–34 species per 4 m^2. There was a significant interaction between treatment area and sampling year for native species richness (Table 2). This was due to an increase in native species richness at the natural regeneration plots and pasture plots through time, but no corresponding increases in the planted plots or the remnant plots (Fig. 4a). Across all years, native species richness was higher in the natural regeneration

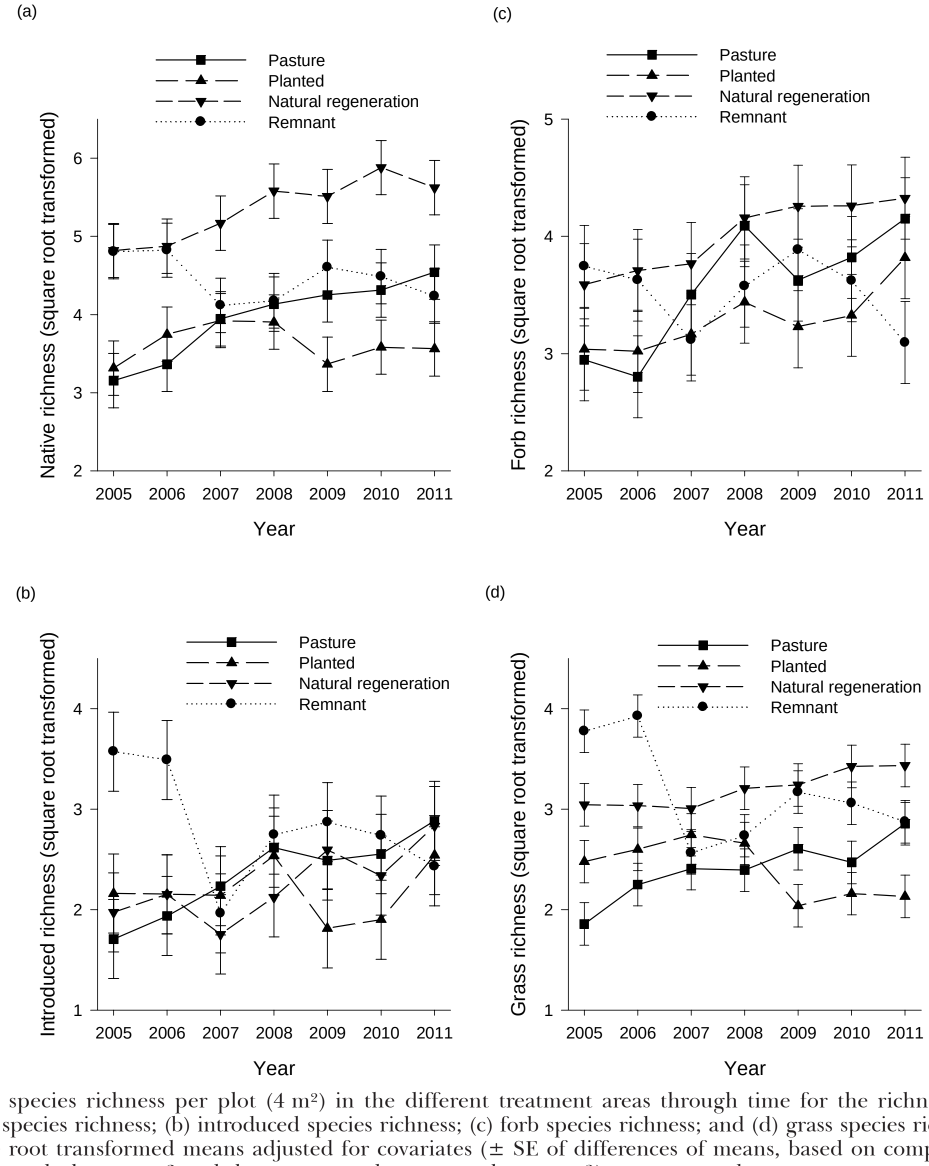

Fig. 4. Plant species richness per plot (4 m^2) in the different treatment areas through time for the richness variables: (a) native species richness; (b) introduced species richness; (c) forb species richness; and (d) grass species richness. Predicted square root transformed means adjusted for covariates (± SE of differences of means, based on comparisons between the planted plots, n = 6 and the pasture and remnant plots, n = 3) are presented.

plots than in the other treatments, but there was no difference between the pasture and planted plots (Fig. 4a). Introduced species richness of the understorey vegetation varied from 0–12 species per plot across all treatment areas and years. There was a significant interaction between treatment area and sampling years for introduced species richness (Table 2). This variable fluctuated somewhat erratically in the remnant plots and planted plots but generally increased over time in the pasture plots and natural regeneration plots (Fig. 4b). Time since fire had an influence on both native and introduced species richness; native and introduced species richness decreased as time since fire increased (Table 2).

There was also a significant treatment area by time interaction for the richness of forbs (Table 2). There was an increase in forb richness in the pasture, planted and natural regeneration plots through time, but temporal fluctuations were observed in the remnant plots (Fig. 4c). Forb richness was higher in the natural regeneration plots than in the planted plots for all years combined (Fig. 4c) and there was a negative relationship between forb richness and litter biomass across all treatments (Table 2). Treatment effects were also inconsistent across years for grass species richness (Table 2). Grass species richness increased gradually through time in the pasture and natural regeneration plots but decreased at the remnant plots between 2006 and 2007 and at the planted plots between 2008 and 2009 (Fig. 4d). Across all times, grass species richness was higher in the natural regeneration and remnant plots than in the pasture and planted plots, which had similar grass richness (Fig. 4d). Grass species richness was negatively related to litter biomass across all years and to time since fire across all treatments (Table 2). The richness of woody plant species (trees and shrubs) varied among treatment areas, but was not influenced by sampling year, or any of the measured covariates (Table 2). The remnant and natural regeneration plots had significantly higher woody plant richness than the pasture and planted plots, which did not differ significantly (5% LSD). Back-transformed means were: 2.0, 1.9, 0.6 and 0.1 plants per species per 4 m^2 for the remnant, natural regeneration, pasture and planted plots, respectively.

All variables had a significant influence on species composition with the exception of topsoil C:N (Table 3). The RDA runs showed that all significant variables explained 47.4% of the variation in species composition. Of that, differences between treatment areas accounted for 27% of the explained variance and other predictor variables accounted for 40.3% of the explained variance in species composition. The confounding effects of treatment area and other environmental variables accounted for 32.7% of the explained variance.

The RDA final model for all sampling years combined identified groups of species associated with the different treatment areas (Fig. 5a). The natural regeneration plots were compositionally distinct from the planted plots. Species associated with the natural regeneration plots included *Emilia sonchifolia*, *Cyanthillium cinereum*, *Hybanthus stellarioides, Crotalaria montana, Themeda triandra*, *Brunoniella australis* and *H. contortus* (Fig. 5a). There were several species that occurred at both the natural regeneration plots and remnant plots that were associated with higher tree basal area, higher litter and

Table 3. Multivariate analyses Monte Carlo results, showing conditional effects and marginal effects on species composition for significant variables. Conditional variance explained is that when all other variables are included in the model, and marginal variance is the amount of variability explained by individual variables (when only that variable is treated as the explanatory variable). Note: a probability of $P = 0.002$ is the lowest achievable given the number of permutations.

Variable	Marginal explained variance (%)	Conditional explained variance (%)	P
Treatment areas:			
(1) Pasture	14.9	10.7	0.002
(2) Planted	14.9	6.4	0.002
(3) Natural regeneration	14.9	17.1	0.002
(4) Remnant	25.6	25.6	0.002
Other variables:			
% fine sand in topsoil	12.8	8.5	0.002
% clay in topsoil	6.4	4.3	0.002
Time since fire	12.8	2.1	0.002
Tree basal area	17.1	6.4	0.002
Herbaceous biomass	4.3	2.1	0.008
Litter biomass	12.8	4.3	0.002
Recent rainfall	6.4	2.1	0.002
Surrounding forest cover	23.5	4.3	0.002
Sampling time	6.4	6.4	0.002

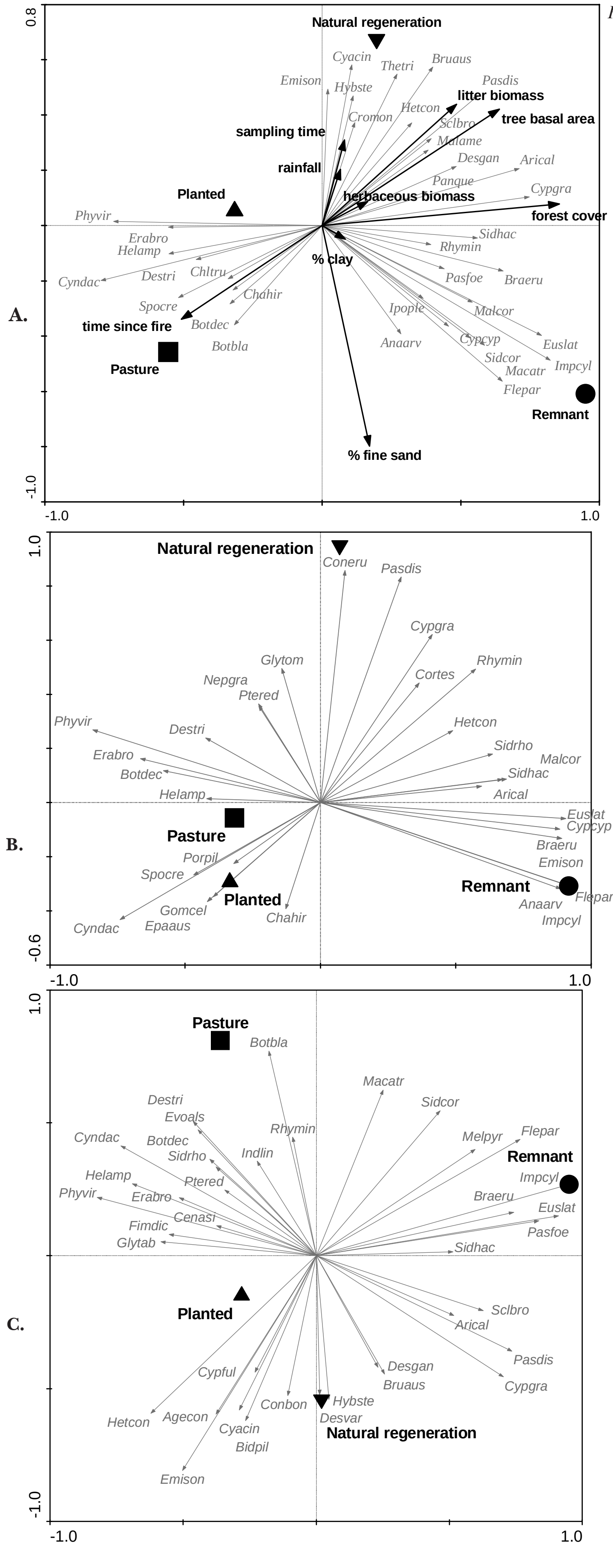

Fig. 5. RDA ordination triplots for: (A) the full model across all times, showing plant species (vectors for abundance scores) associated with the explanatory variables (treatment areas identified by ■, ▲, ▼, ●, sampling time and environmental variables identified by vectors) that had a significant effect on species composition; (B) species associated with the different treatment areas in 2005; and (C) species associated with the different treatment areas in 2011. Only species that show a minimum fit to the model of =15% are displayed. Species abbreviations are: *Emison, Emilia sonchifolia; Cyacin, Cyanthillium cinereum; Hybste, Hybanthus stellarioides; Cromon, Crotalaria montana; Thetri, Themeda triandra; Bruaus, Brunoniella australis; Hetcon, Heteropogon contortus; Pasdis, Paspalidium distans; Sclbro, Scleria brownii; Malame, *Malvastrum americanum; Desgan, Desmodium gangeticum; Panque, Panicum queenslandicum; Arical, Aristida calycina; Cypgra, Cyperus gracilis; Sidhac, Sida hackettiana; Rhymin, Rhynchosia minima; Braeru, *Brachiaria eruciformis; Pasfoe, *Passiflora foetida; Malcor, Malvastrum coromandelianum; Euslat, Eustrephus latifolius; Impcyl, Imperata cylindrica; Cypcyp, Cyperus cyperoides; Sidcor, Sida cordifolia; Macatr, *Macroptilium atropurpureum; Flepar, Flemingia parviflora; Ipople, Ipomoea plebeia; Anaarv, *Anagallis arvensis; Botbla, Bothriochloa bladhii; Chahir, *Chamaesyce hirta; Botdec, Bothriochloa decipiens*; *Spocre, Sporobolus creber; Chltru, Chloris truncata; Destri, *Desmodium triflorum; Cyndac, Cynodon dactylon; Helamp, *Heliotropium amplexicaule; Erabro, Eragrostis brownii; Phyvir, Phyllanthus virgatus*; *Coneru, Convolvulus erubescens; Cortes, Corymbia tessellaris; Sidrho, *Sida rhombifolia; Gomcel, *Gomphrena celosioides; Epaaus, Epaltes australis; Porpil, Portulaca pilosa; Ptered, Pterocaulon redolens; Nepgra, Neptunia gracilis; Glytom, Glycine tomentella; Melpyr; *Melochia pyramidata; Desvar, Desmodium varians; Conbon, *Conyza bonariensis; Bidpil, *Bidens pilosa; Agecon, *Ageratum conyzoides; Cypful, Cyperus fulvus; Glytab, Glycine tabacina; Fimdic, Fimbristylis dichotoma; Cenasi, Centella asiatica; Evoals, Evolvulus alsinoides; Indlin, Indigofera linnaei.*

herbaceous biomass and higher surrounding forest cover and were less abundant in the planted plots and the pasture plots (i.e., *Paspalidium distans, Scleria brownii, *Malvastrum americanum, Desmodium gangeticum, Panicum queenslandicum, Aristida calycina, Cyperus gracilis, Sida hackettiana* and *Rhynchosia minima*). Species associated with the remnant plots included **Brachiaria eruciformis, *Passiflora foetida, Malvastrum coromandelianum, Eustrephus latifolius, Imperata cylindrica, Cyperus cyperoides, Sida cordifolia, *Macroptilium atropurpureum, Flemingia parviflora, Ipomoea plebeia* and **Anagallis arvensis* (Fig. 5a). A smaller group of species were associated with the pasture plots (i.e., *Bothriochloa bladhii, *Chamaesyce hirta, Bothriochloa decipiens, Sporobolus creber* and *Chloris truncata*) and several species were associated with both the pasture and planted plots (**Desmodium triflorum, C. dactylon, *Heliotropium amplexicaule, Eragrostis brownii* and *Phyllanthus virgatus*). Composition in the planted plots was somewhat intermediate

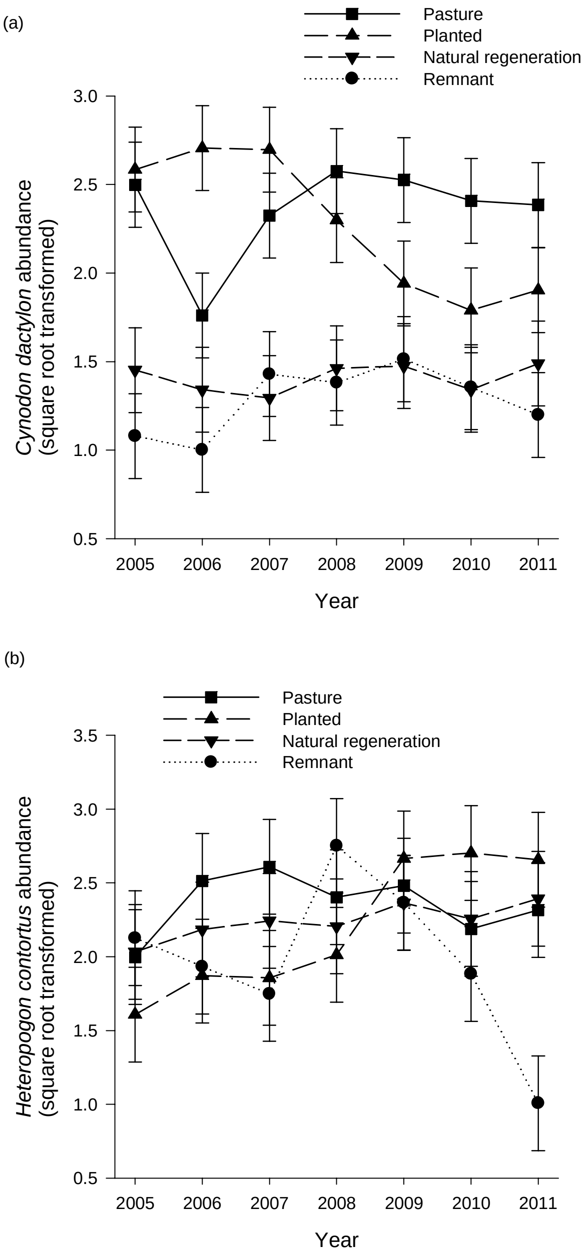

Fig. 6. Cynodon dactylon abundance scores per plot (a); and (b) *Heteropogon contortus* abundance scores per plot in the different treatment areas through time. Predicted square root transformed means adjusted for covariates (± SE of differences of means, based on comparisons between the planted plots (4 m²), *n* = 6 and the pasture and remnant plots, *n* = 3) are presented.

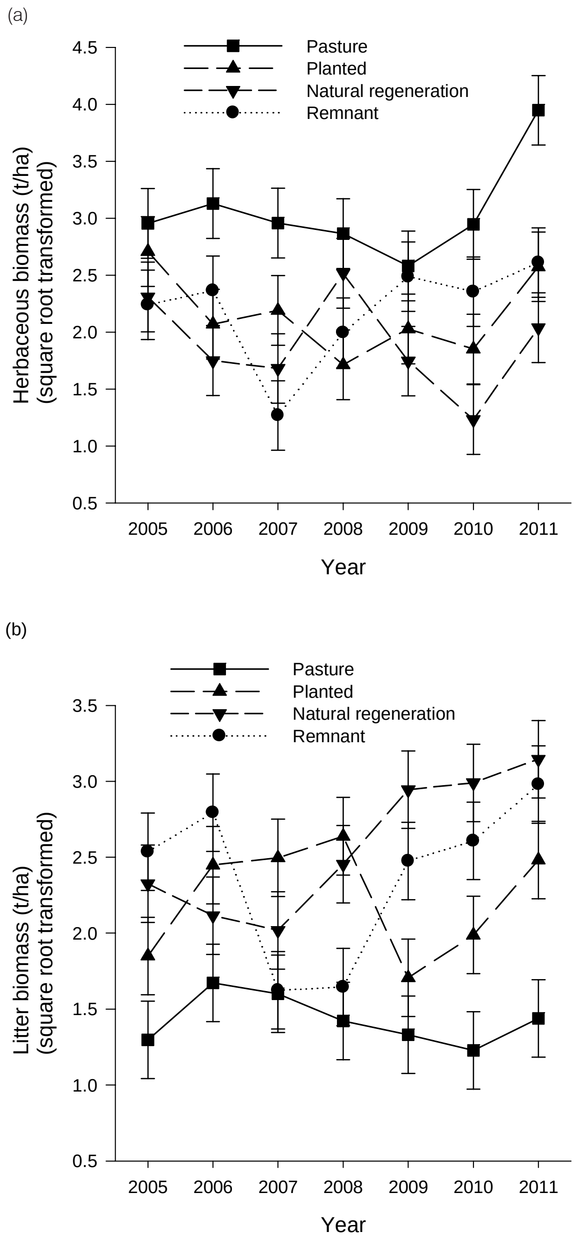

Fig. 7. Herbaceous biomass (a); and (b) litter biomass (t/ha) in the different treatment areas through time. Predicted square root transformed means adjusted for covariates (± SE of differences of means, based on comparisons between the planted plots, *n* = 6 and the pasture and remnant plots, *n* = 3) are presented.

Table 4. Repeated measures ANOVA for the effects of treatment area (pasture, planted, natural regeneration and remnant), time and covariates on herbaceous and litter biomass and topsoil percentage C and N between 2005 and 2011. *P <0.05, **P <0.01, ***P <0.001. NA, not applicable.

Source of variation	Herbaceous biomass	Litter biomass	% C	% N
plot stratum				
Treatment	*ns*	$F_{3,12}$ = 17.45***	$F_{3,6}$ = 8.10*	*ns*
Covariates	$F_{4,7}$ = 7.36*	*ns*	$F_{5,6}$ = 11.29**	$F_{4,7}$ = 8.85**
% clay	*ns*	*ns*	$F_{1,6}$ = 19.19**, β = 2.94	$F_{1,7}$ = 16.36**, β = 3.00
% fine sand	*ns*	*ns*	$F_{1,6}$ = 13.42*, β = 3.18	$F_{1,7}$ = 10.37*, β = 3.36
C:N	*ns*	*ns*	NA	NA
Rainfall	NA	NA	*ns*	*ns*
Tree basal area	$F_{1,7}$ = 7.52*, β = 3.65	*ns*	$F_{1,6}$ = 18.96**, β = 3.71	$F_{1,7}$ = 8.30*, β = 2.65
Herbaceous biomass	NA	NA	*ns*	*ns*
Litter biomass	NA	NA	*ns*	*ns*
Time since fire	$F_{1,7}$ = 12.00*, β = –3.46	*ns*	*ns*	*ns*
Plot.Time stratum				
Time	$F_{6,63}$ = 6.87***	$F_{6,75}$ = 6.46***	*ns*	$F_{6,63}$ = 4.02**
Time×Treatment	$F_{18,63}$ = 6.18***	$F_{18,75}$ = 8.19***	$F_{18,63}$ = 5.11***	$F_{18,63}$ = 4.23***
Covariates	*ns*	*ns*	$F_{2,63}$ = 3.77*	*ns*
C:N	*ns*	*ns*	NA	NA
Tree basal area	*ns*	*ns*	$F_{1,63}$ = 4.13*, β = 2.02	*ns*
Herbaceous biomass	NA	NA	*ns*	$F_{1,63}$ = 4.79*, β = 2.19
Litter biomass	NA	NA	*ns*	*ns*
Time since fire	*ns*	*ns*	*ns*	*ns*

between the pasture plots and the natural regeneration plots, but differed greatly to that of the remnant plots (Fig. 5a).

The composition in the planted plots was more similar to that of the pasture plots than to that of the natural regeneration and remnant plots for all years combined (Fig. 5a). However, understorey species composition varied through time (Table 3). In 2005, the planted and pasture plots had a similar composition (R = –0.24, P >0.05) but composition of the planted plots differed significantly to that at the natural regeneration and remnant plots (R = 0.57, P <0.05 and R = 0.96, P <0.05, respectively; Fig. 5b). However, by 2011 there was significant separation between the planted and pasture plots in terms of species composition (R = 0.82, P <0.05; Fig. 5c). There were trends to suggest some change in composition in the planted plots over time for two common grass species. There were significant treatment by time interactions for both *Cynodon dactylon* and *Heteropogon contortus* (Table 2). *Cynodon dactylon* decreased in abundance through time in the planted plots, but did not decrease consistently in the other treatment areas, while *H. contortus* abundance increased somewhat in the planted plots through time, but fluctuated significantly in the remnant plots or did not vary greatly in the pasture or natural regeneration plots (Fig. 6).

Herbaceous and litter biomass

There were significant treatment by time interactions for herbaceous and litter biomass (Table 4). There was a decrease in biomass in both cases between 2006 and 2007 in the remnant plots that was not apparent in the other treatments (Fig. 7) that was most likely attributable to the fire at these plots prior to sampling in 2007. There was also a decrease in litter biomass between 2008 and 2009 in the planted plots that was not apparent in the other treatments (Fig. 7), which is also likely related to the prescribed burning that took place in this treatment area in September 2008. Across all times, litter biomass was significantly lower in the pasture plots than in all other treatments (Fig. 7b). There was a positive relationship between tree basal area and herbaceous biomass, and herbaceous biomass increased as time since fire increased, for all years combined (Table 4).

Soil carbon and nitrogen

Percentage C in the topsoil varied from 1.2–3.8% and total N varied from 0.10–0.25% for all treatment areas and sampling years combined. There were significant time by treatment interactions for both topsoil C or N (Table 4); this was due to increases in both C and N between 2009 and 2011 in the planted plots but decreasing trends in the other treatment areas (Fig. 8). Across all times, topsoil C was higher in the natural regeneration plots than in all other treatments (means adjusted for covariates of 2.50% in the natural regeneration plots, 1.58%, 1.75% and 1.87% in the pasture, remnant and planted plots, respectively, LSD =

0.73). Across all times, percentage clay, fine sand, and tree basal area were positively associated with topsoil C. The relationship with tree basal area and topsoil C was also positive for all treatments combined (Table 4). Topsoil N was positively related to percentage clay, fine sand and tree basal area across all times, while there was a positive relationship between topsoil N and herbaceous biomass across all treatments (Table 4).

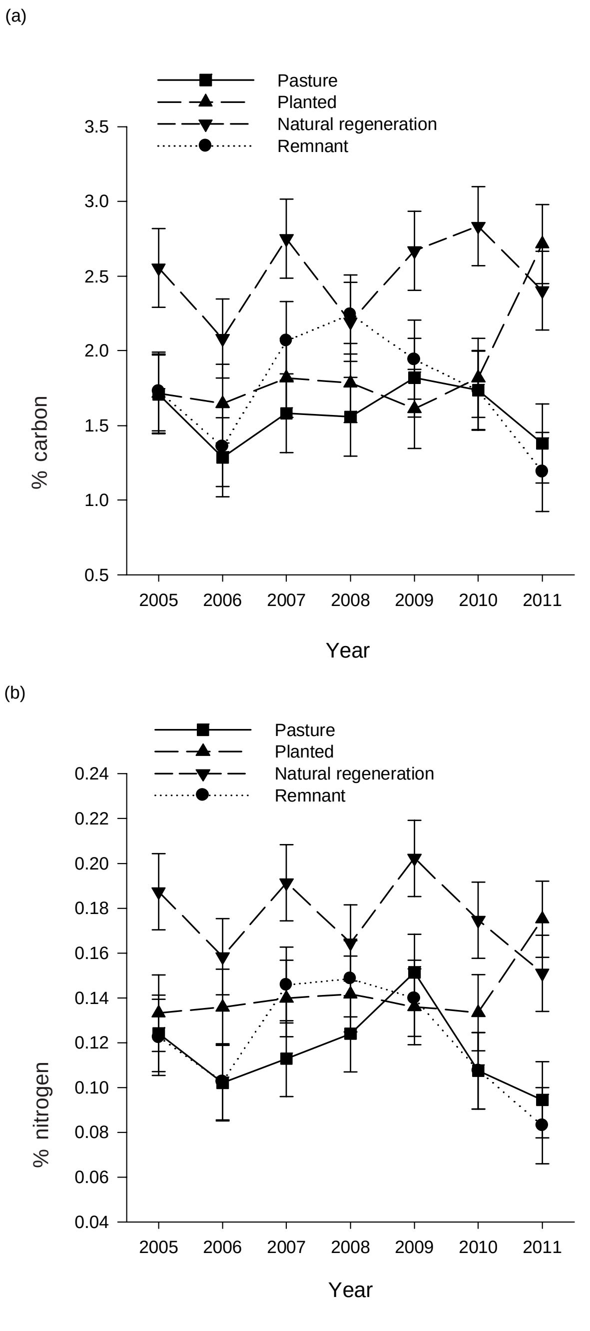

Fig. 8. Percentage carbon in the topsoil (a); and (b) % nitrogen in the topsoil in the different treatment areas through time. Predicted means adjusted for covariates (± SE of differences of means, based on comparisons between the planted plots, n = 6 and the pasture and remnant plots, n = 3) are presented.

DISCUSSION

Is the revegetation site diverging from cleared pasture?

We found evidence that the revegetation site is diverging from cleared pasture in terms of understorey species composition, vegetation structure and topsoil C and N in the nine years since revegetation efforts began. Understorey plant species composition appears to be following a gradual, but positive restoration trajectory as the planted plots initially had a similar composition to the pasture plots but these treatments diverged in composition over time. The transition in the planted plots is demonstrated through the decreased abundance over time of *C. dactylon,* a common pasture grass, while *H. contortus,* a common grass that occurs naturally in open forest and woodlands in the region, increased in cover. Despite such changes in plant composition there were still significant differences in composition between the planted plots and plots in the treatments that had not been heavily cleared (i.e., the natural regeneration and remnant plots). In fact, there was no convergence in composition between the planted plots and remnant plots over time. This is not surprising as it is likely that some native plant species have been eliminated from the area or have had their abundance reduced due to the history of clearing and cattle grazing (Kaur *et al*. 2006; Standish *et al*. 2007; Lewis *et al*. 2009) and the associated changes in soil properties, such as loss of soil structure (Yates *et al*. 2000; Standish *et al*. 2006; Flinn and Marks 2007). Similar findings were reported by Wilkins *et al*. (2003) and Nichols *et al*. (2010) for Cumberland Plain woodlands, near Sydney; where tree planting had little impact on native understorey species assemblages for plantings up to 10 years of age. In these cases, and in the current study, it is likely that seed limitation is a major factor influencing re-establishment of a native understorey composition. Hence, restoration of open forests and woodlands with a mostly herbaceous understorey plant composition is likely to require seed dispersal (e.g., by wind, ants, other animals or humans) from the surrounding forest (Yates and Hobbs 1997) and could be hastened through sowing of native understorey seed (e.g., Gibson-Roy *et al*. 2007).

There was evidence of changes in vegetation structure over time to support the hypothesis that the revegetation site is diverging from cleared pasture. There was an increase in the density of saplings over time in the natural regeneration plots, and the planted plots had higher sapling densities than the pasture plots, as would be expected following planting. Our findings suggest that, despite a history of livestock grazing, there is potential for natural regeneration in this ecosystem and such potential also exists in similar grassy forest and woodlands elsewhere (Dorrough and Moxham 2005). Development of regeneration to saplings in the natural regeneration and planted plots suggests that removal of grazing has been important for allowing this recovery, given that similar development of regeneration did not occur in the pasture plots. While high levels of regeneration persisted in one of the pasture plots through time, this regeneration remained stunted (no trees had a DBH of >2 cm in the pasture plots), presumably due to the effects of browsing. Observations suggest that most of the tree species regenerating in this ecosystem are able to produce lignotubers, as many of the individuals 0.5–1 m in height that were tagged in 2005 were still alive in 2011 despite the occurrence of fire and grazing at some plots. Such a regeneration pool of small trees (usually <1 m in height) can persist in a dormant state for long periods of time (e.g., more than 15 years in some cases; Florence 1996) waiting for improved conditions for development (e.g., release from a competing overstorey or from browsing pressure). Hence, some period of grazing and fire exclusion may benefit the development of lignotuberous regeneration in *E. tereticornis* forest, and is known to encourage tree regeneration elsewhere (e.g., Spooner *et al*. 2002; Dorrough and Moxham 2005; Briggs *et al*. 2008).

There was weak evidence of divergence in topsoil C and N between the planted plots and the pasture plots (trends of an increase in these variables between 2009 and 2011 in the planted plots but a decrease in the pasture plots). This trend suggests that despite the history of more intensive agriculture across the planted area and the relatively short period of time since revegetation, natural soil processes may be occurring in these areas as the planted trees develop (e.g., through development of root systems and providing nutrients in leaf litter; Rhoades *et al*. 1998; Paul *et al*. 2002; Prober *et al*. 2002a). Potentially greater accumulation of soil carbon would have been observed in the planted area if we had sampled to greater depths (i.e., >10 cm depth) and included the >2 mm soil carbon fraction, to increase the probability of sampling tree root fragments. Reviews suggest that there is considerable variation in the rates of soil carbon accumulation following reforestation and afforestation (Post and Kwon 2000; Guo and Gifford 2002; Paul *et al*. 2002). Given that there are few studies of similar ecosystems to allow comparison of changes in soil carbon and nitrogen over time following reforestation (e.g., Post and Kwon 2000), future monitoring of soil attributes at the established plots is recommended.

Evidence of a divergence between the revegetation site and the pasture was not strong for woody plant regeneration density, plant species richness and ground layer biomass. This was partly due to the fact that there were not large differences in these variables between the treatment areas when monitoring began in 2005. Interestingly, there were no significant differences in the richness of native and introduced species, forb and woody plant species richness, herbaceous biomass and woody plant regeneration density between the pasture plots and the remnant plots after accounting for the covariates that were measured. While the pasture plots chosen were representative of the revegetation site prior to revegetation in terms of soil type and position in the landscape, grazing intensity at the pasture plots between 2005 and 2011 has not been as high as it was at the revegetation site prior to revegetation efforts (e.g., Appendix 1) and records suggest the pasture plots were not overgrazed historically. The moderate grazing intensity and periodic nature of cattle grazing at the pasture plots may have contributed to the high density of woody plant regeneration in one of the three randomly positioned plots that was located nearby an isolated patch of mature paddock trees (Dorrough and Moxham 2005). Ideally a greater number of pasture plots with differing management histories would have been sampled as it is possible that the three plots sampled are unrepresentative of cleared and grazed *E. tereticornis* plains elsewhere in the region, in terms of woody plant regeneration.

There was no divergence in native species richness between the natural regeneration plots and pasture plots, as in both cases richness increased over time. Divergence in native plant richness between the planted plots and pasture plots through time was in the opposite direction to that expected; because there was no similar increase in richness the planted plots. Thus, despite some compositional change in the planted plots it appears that removal of certain native species through clearing and grazing may have a longer-term impact on plant diversity. For example, across all times, the richness of all grass species was lower in the planted and pasture plots. Thus removal of livestock grazing from agricultural land is unlikely to result in an immediate recovery of plant diversity where conditions or resources have been altered through past management (Yates *et al*. 2000).

Factors influencing *E. tereticornis* community assemblages

Understorey plant species composition varied between the treatment areas and was influenced by site-related factors (tree basal area, soil texture, ground layer biomass, time since fire, recent rainfall), landscape factors (surrounding forest cover) and time. Similar factors also influence understorey compositions in other Australian grassy woodlands (e.g., McIntyre *et al.* 2003; Scott *et al.* 2009). In the current study a large degree of the explained variation in plant species composition (32.7%) was due to the confounding effects of treatment and the other measured variables. This is not surprising given the management history differences between the treatment areas, and the likely influence of land use history on understorey composition (Foster *et al.* 2003; Lunt and Spooner 2005). Nevertheless, results suggest that development of herbaceous and litter biomass, tree basal area and surrounding forest cover over time should encourage further compositional changes in the planted plots so that this area becomes more similar to the natural regeneration and remnant areas. Thus future changes in species composition will be affected by changes in the environmental characteristics of the revegetation site. For example, the positive relationship between topsoil fertility and tree basal area suggests that further development of basal area will increase topsoil C and N over time and this is likely to encourage compositional change. However, species composition of the revegetation site will probably always differ to that of the remnant site to some degree given the differences in soil texture between treatments (i.e., higher percentage clay and fine sand in the topsoil).

Richness of native, introduced, grass and forb species was influenced by certain covariates. As time since fire increased native and introduced species richness decreased. This was driven directly by a negative relationship between grass species richness and time since fire and indirectly through negative relationships between forb species richness and litter biomass and grass species richness and litter biomass. This has important implications for the management of this ecosystem as it suggests that herbaceous plant diversity is encouraged by disturbance that reduces phytomass, such as fire. Similar findings have been reported for other grassy ecosystems in eastern Australia (e.g., Lunt and Morgan 2002; Schultz *et al.* 2011). Managers attempting to restore *E. tereticornis* forest ecosystems on former agricultural land should therefore consider implementing a fire frequency regime that will encourage herbaceous diversity but still allow development of lignotuberous woody plants in the period between fires. However, care should be taken to ensure such a fire management regime does not encourage introduced species, where introduced species are common.

Conclusions

Despite some divergence between the revegetation site and the pasture, monitoring over a long time-frame will be necessary to determine the success of this revegetation given the time required for establishment of forest cover and natural ecosystem processes (e.g., dispersal of understorey plant species) in heavily cleared parts of the landscape. Due to the time lag between vegetation clearing and re-establishment of forest cover following a vegetation offset agreement, and the rarity of intact and mature *E. tereticornis* communities on alluvial plains, offset agreements should consider options other than revegetation where possible. Ideally, conservation of *E. tereticornis* ecosystems should involve protection of intact remnants in combination with management and protection of areas that have not been heavily cleared and overgrazed, as there is a high potential for recovery of woody vegetation cover in this ecosystem through natural regeneration. Nevertheless, the revegetation methodology described here to establish a *E. tereticornis* plantation forest is considered successful and could be applied to similar areas of cleared former *E. tereticornis* forest or woodland.

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APPENDIX 1.

Photographs showing: (a) the revegetation site in 2001 prior to planting; (b) a planted revegetation plot in 2005; (c) a pasture plot in 2005; (d) a planted revegetation plot in 2010; (e) a pasture plot in 2010, showing dense regeneration <1 m in height.

a

b

c

d

e