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THE EFFECT OF TREE DENSITY AND VIGOROUS LEGUMES ON UNDERSTOREY SPECIES RICHNESS, DENSITY AND COVER IN REHABILITATED BAUXITE MINES

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SUMMARY

The target tree establishment density for rehabilitated mine pits in the jarrah forest is currently 1300 stems ha^{-1} and is designed to ensure that 300 stems ha^{-1} of sawlogs are available for future timber production. However, tree density may affect other ecosystem parameters such as water use and understorey species richness. Rehabilitated sites were historically seeded with an understorey mix that included vigorous legumes to establish rapid vegetation cover and fix atmospheric nitrogen. Similar to high tree density, these vigorous legumes may have the undesirable effect of out-competing the small understorey species, which are the main contributors to the species richness of rehabilitated mined sites. The major objectives of this study were to determine 1) if high tree densities reduce the richness, density and cover of understorey species and 2) if the competitive effect of a vigorous native legume seed mix on smaller understorey species is evident 15 years post-rehabilitation. A complete factorial experiment of four tree densities (625, 1250, 2500 or 10 000 stems ha^{-1}) and two seed treatments (a high rate of vigorous native legume seed [+L] and no vigorous legume seed [-L]) were established in five replicate sites in 1988 in rehabilitated mined areas. Understorey species richness, density and cover were measured in a total of 80 m^2 in each of the 40 treatment areas in spring 2003. The applied seed mix (+L or –L) had a significant effect on the understorey vegetation, while the effect of tree density was less pronounced. Total species richness was significantly lower in +L (25.5 plants/80 m^2) compared with –L (33.2 plants/80 m^2) seeded sites, while total plant cover was significantly higher in +L (66%) compared with –L (44%) seeded sites. The density and/or cover of several small native understorey species and genera, some of which were recalcitrant species, were all significantly higher in –L compared with +L seeded sites. The species richness of groups of species, such as monocots, small non-leguminous shrubs and recalcitrant species, were also significantly higher in –L compared with +L seeded sites. Tree density affected the species richness of monocots, recalcitrant species and the cover of one individual species (*Opercularia echinocephala*). Generally, reductions were seen at 10 000 stems ha^{-1} compared with 625 - 2500 stems ha^{-1}. We conclude that tree densities within the acceptable range for post-1988 rehabilitated sites (600 – 2500 stems ha^{-1}) will not significantly impact the richness, density or cover of understorey established in rehabilitated sites. The application rate of vigorous legume seed has

been progressively reduced over time from 2 kg ha^{-1} in 1988 to less than 0.4 kg ha^{-1} in 2005 based on the results of this and other research.

INTRODUCTION

Alcoa World Alumina Australia (Alcoa) mines and rehabilitates approximately 550 ha annually in the northern jarrah (*Eucalyptus marginata* D.Don ex Sm.) forest of Western Australia. The objective of rehabilitation is to re-establish a self-sustaining jarrah forest ecosystem designed to enhance or maintain water, timber, recreation, conservation and other nominated forest values (Elliott *et al.* 1996; Gardner 2001). One of Alcoa's completion criteria states that rehabilitated sites must be capable of producing 300 stems ha^{-1} of sawlogs at 15 years of age (DoIR 2002). The current rehabilitation prescription has a target tree density of 1300 stems ha^{-1} at nine months of age, with a maximum stocking rate of 2500 stems ha^{-1} and minimum of 600 stems ha^{-1}. Maximum and minimum stocking rates were specified because jarrah establishment in one-year-old rehabilitated sites has been variable despite a constant seed application rate. For instance, when jarrah is broadcast at a rate of 2.5 kg ha^{-1}, establishment ranges from 1104 – 5829 stems ha^{-1} (Koch and Ward 2005). The eucalypt seed application rate has been progressively reduced over time to 1.9 kg ha^{-1} in 1995 and lowered again to 0.8 kg ha^{-1} in 2003, due to concerns that high tree density may have silvicultural effects such as decreased tree height and stem diameter (Koch and Ward 2005) and increased water use (Croton 2004).

Understorey establishment may also be affected by the tree stocking rate due to competition for light, water and nutrients. The principles of ecologically sustainable management for the jarrah forest (Conservation Commission of WA 2004) mean that rehabilitated areas must fulfil a range of future land uses, including the production of sawlogs and the return of a species rich flora to restore conservation values. To achieve maximum species richness, the topsoil is returned, native seed is broadcast (70 – 100 species) and greenstock of species known to be difficult to re-establish are planted. Alcoa's current internal target is to establish 100% of the indigenous plant species found in representative jarrah forest sites in 15-month-old rehabilitated areas, with 20% of these species from the 'difficult to re-establish' group. In rehabilitated coal mines in the USA, the rapid increase in overstorey cover of *Pinus strobus* in order to meet tree establishment requirements affected the herbaceous understorey species. Most understorey species were found in gaps between trees, with species richness expected to fall as the canopy matures and closes (Holl 2002). In rehabilitated jarrah forest, there may be an effect of tree stocking rate on understorey species richness, which would be an important consideration in attempting to fulfil the multiple land-use values of the rehabilitated jarrah forest.

In 1988, Alcoa's broadcast seed mix consisted of around 14 species of vigorous native legumes (fast-growing legumes and acacias) and approximately 80 small shrub and herb species. The vigorous native legumes were the main contributor to initial plant cover and biomass in rehabilitated areas, while the smaller shrub and herb species and seed from the returned topsoil were the main contributors to plant diversity. While the vigorous native legumes provide rapid vegetation cover to prevent erosion, fix atmospheric nitrogen, create litter and build up soil organic carbon, if they are too dense they reduce jarrah tree growth, create a fire hazard by providing a large amount of aerated fuel, reduce access to rehabilitated areas, and give a structure and appearance unlike that of the surrounding unmined forest (Koch and Ward 2005). In addition, Koch and Davies (1993) found significantly fewer small shrub and herb species and reduced diversity in areas with dense legume vegetation compared with sparse areas. The vigorous early growth of the tall acacias appears to out-compete the smaller species even in the first year after rehabilitation, with a significant reduction in the number of understorey species still noted 9 to 12 years later when most of the acacias have senesced (Koch and Davies 1993). The longer-term effect of sites initially seeded with a vigorous legume understorey on the smaller understorey species is unknown.

The rate of application of seed of vigorous legume species was reduced from 2 kg ha^{-1} in 1988 to less than 0.4 kg ha^{-1} in 2005 (Koch and Ward 2005), to reduce some of their negative impacts. It is important that maximal species richness is attained in the first year of growth in rehabilitated sites, because successional development appears to be controlled by the group of species initially establishing on the site (Koch and Ward 1994; Norman *et al*. 2006). The major objectives of this study were to 1) determine if high tree densities reduce the richness, density and cover of understorey species and 2) determine if the competitive effect of a vigorous native legume seed mix on smaller understorey species is evident 15 years post-rehabilitation.

METHODS

Five replicate sites were established at four mines in late autumn 1988. There was one site at Jarrahdale (Peacock), one at Huntly (Possum), two at Del Park (Karri and Scribbly) and one at Willowdale (Scarp North) mines. A factorial combination of four planted jarrah tree densities (625, 1250, 2500 or 10 000 stems ha^{-1}, or 4 x 4, 2 x 4, 2 x 2 or 1 m x 1 m spacing) and two seed treatments (vigorous native legume seed mix [+L] and no vigorous native legume seed mix [-L]) were applied in a split-plot experimental design across the five sites. Each treatment consisted of 10 rows of 10 trees, resulting in treatment areas of 40 x 40, 20 x 40, 20 x 20 or 10 m x 10 m as the tree density increased. The vigorous native legume seed mix, applied at 2 kg ha^{-1}, consisted of 14 plant species, predominantly acacias and other legumes that are fast growing and relatively tall shrubs (2-4 m). All sites were seeded with the basal understorey seed mix, applied at 0.5 kg ha^{-1}, of 80 species that are small shrubs and herbs usually less than 0.5 m tall. Each plot was fertilised with 500 kg ha^{-1} of superphosphate (41 kg ha^{-1} P, 3.3 kg ha^{-1} Cu, 1.5 kg ha^{-1} Zn, and 0.2 kg ha^{-1} Mo) and 250 kg ha^{-1} diammonium phosphate (43.75 kg ha^{-1} N and 50 kg ha^{-1} P). At the north-west corner of each of the 40 treatment areas, a rectangular plot (10 m x 8 m) was established to monitor the understorey vegetation. Twenty 2 m x 2 m quadrats (a total of 80 m^2) were established within the plot and the density and cover of each plant species recorded. Monitoring occurred in spring 2003, 15 years post-establishment.

All species were identified according to the nomenclature of Florabase (2006). Non-native species (weeds) were separated from native species for all univariate vegetation analyses except total plant density and cover. Shannon-Wiener diversity (H′) and evenness (E) were calculated using density values for each site with the following equations:

$$H' = - \sum(p_i) \ln (p_i)$$

where p_i = the proportion of the ith species.

$$E = H' / H_{max}$$

where $H_{max} = \ln(s)$ and s = the total number of species.

Analysis of treatment effects was undertaken on individual species, genera and species groups of interest. Only species with density and cover in 10 or more of the

40 plots were included in the analysis of individual species (52 of 192 live and dead species). Species were grouped into 30 native and one non-native genera (minimum of two species per genera). Recorded species were also divided into groups of interest namely 'recalcitrants' (41 species), monocots (46 species), small non-leguminous shrubs (39 species), small leguminous shrubs (14 species), ephemerals (25 species) and orchids (13 species). Alcoa has defined recalcitrant species as those that are difficult to establish and are common in the forest but absent or found in low densities in rehabilitated mine sites. There were a few instances of species overlap between groups, for instance a monocot could also be a recalcitrant species.

Vegetation analyses were undertaken only on live plant density and cover data. The exception was the individual species analyses, where dead plant density and cover were also analysed. Dead plant cover, particularly of the large legumes that that formed the +L seed mix, contributes to the fire risk of rehabilitated sites. There was high density and cover of the ephemeral weeds *Aira caryophyllea* and *Lotus uliginosus* in the -L seeded, 2 m x 4 m tree spacing at the Scribbly site, which were removed prior to data analysis. There was also a high density of *A. caryophyllea* in the +L and -L seeded, 2 m x 2 m tree spacing at the Peacock site, which was removed prior to data analysis.

Univariate analyses of treatments were performed using Analysis of Variance (ANOVA) and Regression Analysis in Minitab 12.12®. The ANOVA model used tree density as the main-plot treatment, seed as the split-plot treatment and site as a random variable. There were no significant interactions between factors. The Regression Analysis used tree density as the predictor factor. Prior to analysis, all data exhibiting non-normal distributions or unequal variances were square root, cubic root or log transformed. In instances where the data continued to exhibit non-normal distributions or uneven variances regardless of the applied transformation, the more conservative non-parametric Kruskal-Wallis test for significance was used. Data were also analysed without the 10 000 stems ha^{-1} treatment to try and elucidate differences between the lower tree densities (625 - 2500 stems ha^{-1}), however there were no significant differences.

Multivariate analyses were undertaken on understorey density and cover data, including weed species. The same outlying ephemeral weed species that were removed prior to univariate analysis were removed prior to multivariate analysis. Species ordinations are not presented due to the large number of points. A two-way Analysis of Similarity (ANOSIM) of density and cover data based on Bray-Curtis similarity matrices were conducted using PRIMER 6. The application used a maximum of 9999 permutations to determine significant differences between the experimental factors. Multi-dimensional scaling (MDS) ordinations of significant factors were then performed.

RESULTS

Univariate analyses

Tree density did not significantly affect any of the five vegetation parameters (Table 1). The seed treatment significantly affected total plant cover and native species richness, while site significantly affected three of the vegetation characteristics. Total plant cover was significantly higher (P = 0.01) in +L (66%) compared with -L (44%) seeded sites. In contrast, native species richness was significantly higher (P = 0.02) in -L (33.2 plants/80m^2) compared with +L (25.5 plants/80m^2) seeded sites. There was a non-significant trend (P = 0.085) of lower plant density in +L (5.2 plants/m^2) compared with -L (7.5 plants/m^2) seeded sites.

Table 1: A summary of the statistical significance of differences between the various combined vegetation parameters and site (Karri, Peacock, Possum, Scarp or Wriggly), tree density (625, 1250, 2500 or 10 000 stems ha^{-1}) and seed treatment (vigorous native legumes [+L] or no vigorous native legumes [-L]). NS = P > 0.05, * = P ≤ 0.05, ** = P ≤ 0.01. There were no factor interactions (n = 40).

Vegetation Parameter	Site	Tree Density	Seed
Total plant density/m^2	*	NS	NS
Total plant cover	NS	NS	*
Native species richness/80m^2	**	NS	*
Native species diversity	*	NS	NS
Native species evenness	NS	NS	NS

There was a non-significant trend (P = 0.07) of lower native species richness at high compared with lower tree densities (Table 2). At 10 000 stems ha^{-1}, there were 22.1 native species/80 m^2 compared with 26.6 – 34.8 native species/80 m^2 at 625 – 2500 stems ha^{-1}. At 10 000 stems ha^{-1} there were 2.1 plants/m^2 and 44% cover compared with 6.3 – 9.0 plants/m^2 and 57 – 60% cover at 625 – 2500 stems ha^{-1} (Table 2).

Table 2: Native species richness, total understorey density and cover at four tree densities (625, 1250, 2500 or 10 000 stems ha^{-1}). Numbers are means ± S.E. Differences between tree densities treatments were non-significant ($P > 0.05$). For each value, n = 10.

	Tree density (stems ha^{-1})			
	625	**1250**	**2500**	**10 000**
Native species richness/80m^2	33.9 ± 3	26.6 ± 5	34.8 ± 6	22.1 ± 3
Density (plants/m^2)	8.1 ± 2	6.3 ± 3	9.0 ± 3	2.1 ± 0
Cover (%)	59.7 ± 9	57.2 ± 10	57.1 ± 8	44.4 ± 4

A total of 158 live native and 21 weed species were identified across all sites. Dead plant density and cover were recorded for 13 species. Tree density did not significantly affect the density or cover of individual understorey species or genera. The exception was *Opercularia echinocephala*, which showed significantly increased cover ($P = 0.043$) as the tree density decreased, averaging 0, 0.04, 0.08 and 0.12% cover at 10 000, 2500, 1250 and 625 stems ha^{-1} respectively. Cover of *O. echinocephala* was significantly lower at 10 000 and 2500 compared with 625 stems ha^{-1}.

The seed treatment significantly affected the density and/or cover of 12 live native, two live weed and two dead species (Table 3). Density and/or cover of the four live and two dead *Acacia* species that were part of the vigorous native legume seed mix were higher in +L compared with -L seeded sites. Density and/or cover of the three recalcitrant species *Boronia fastigiata*, *Lasiopetalum floribundum* and *Lomandra sonderi* were higher in -L compared with +L seeded sites. Density and/or cover of the remaining five native (*Bossiaea aquifolium, Hypocalymma angustifolia*, *Opercularia echinocephala*, *Phyllanthus calycinus* and *Senecio quadridentatus*) and two weed (*Disa bracteata* and *Hypochaeris glabra*) species were also higher in -L compared with +L seeded sites.

Table 3: Density (plants/m^2) and cover (%) of individual species based on seed treatment (vigorous native legumes [+L] or no vigorous native legumes [-L]). Signif. = Significance (NS = P > 0.05, * = P < 0.05, ** = P < 0.01 and *** = P < 0.001). Only species showing significant differences are presented. For each value, n = 20.

Species	**Density (plant/m^2)**		**Signif.**	**Cover (%)**		**Signif.**
	+L	**-L**		**+L**	**-L**	
Live						
Acacia drummondii	0.04 a	0.01 b	*	1.57 a	0.19 b	*
Acacia lateriticola	0.04 a	0.01 b	*	2.04 a	0.18 b	*
Acacia longifolia	0.11	0.06	NS	40.5 a	8.90 b	***
Acacia urophylla	0.01 a	0.00 b	*	0.25	0.02	NS
Boronia fastigiata	0.00 b	0.03 a	**	0.00 b	0.05 a	*
Bossiaea aquifolium	0.60 b	1.00 a	*	9.07 b	18.4 a	*
Disa bracteata (weed)	0.02	0.11	NS	0.00 b	0.04 a	*
Hypocalymma angustifolia	0.01 b	0.03 a	*	0.30	1.30	NS
Hypochaeris glabra (weed)	0.17 b	0.30 a	*	0.04 b	0.14 a	*
Lasiopetalum floribundum	0.03 b	0.07 a	**	1.34	1.27	NS
Lomandra sonderi	0.00	0.20	NS	0.00 b	0.06 a	*
Opercularia echinocephala	0.06 b	0.11 a	*	0.02 b	0.10 a	**
Phyllanthus calycinus	0.01 b	0.04 a	*	0.10	0.25	NS
Senecio quadridentatus	0.00 b	0.02 a	**	0.00 b	0.05 a	*
Dead						
Acacia celastrifolia	0.02 a	0.00 b	*	4.20 a	0.17 b	*
Acacia longifolia	0.07	0.01	NS	5.80 a	1.04 b	**

The seed treatment significantly affected the density and/or cover of eight native genera (Table 4). The *Acacia, Bossiaea, Lomandra* and *Opercularia* genera consisted of species that were significant as individuals. There were four new significant genera; *Caladenia, Dryandra, Gompholobium* and *Hibbertia*. There was significantly higher density of the *Caladenia* (Orchidaceae), *Hibbertia* (two of the four *Hibbertia* species were recalcitrants) and *Dryandra* genera in –L compared with +L seeded sites. There was significantly higher density and cover of the *Gompholobium* genus in –L compared with +L seeded sites.

Table 4: Density (plants/m^2) and cover (%) of genera based on seed treatment (vigorous native legumes [+L] or no vigorous native legumes [-L]). Signif. = Significance (NS = P > 0.05, * = P < 0.05, ** = P < 0.01 and *** = P < 0.001). Only genera showing significant differences are presented. For each value, n = 20.

Genera	Density (plants/m^2)		Signif.	Cover (%)		Signif.
	+L	-L		+L	-L	
Acacia	0.44	0.18	NS	46 a	12 b	***
Bossiaea	0.64 b	1.11 a	**	1.84 b	3.04 a	*
Caladenia	0.07 b	0.10 a	*	0.01	0.02	NS
Dryandra	0.00 b	0.04 a	*	0.25	0.97	NS
Gompholobium	0.00 b	0.02 a	*	0.01 b	0.02 a	*
Hibbertia	0.01 b	0.04 a	*	0.01	0.06	NS
Lomandra	0.02 b	0.05 a	*	0.06 b	0.12 a	**
Opercularia	0.07 b	0.15 a	*	0.03 b	0.13 a	*

The cover of the *Acacia* genus was 46% in sites seeded with a vigorous legume seed mix compared to 12% when vigorous legumes were not seeded (Table 4). However, it appears that the *Bossiaea aquifolium* had a compensatory effect because cover increased from 9.6 to 19.1% when the vigorous legumes were not seeded. *Bossiaea aquifolium* is a large seeded legume, however it also establishes well from the topsoil seed store. Although non-significant, there was a trend of increasing cover of *B. aquifolium* as the tree density reduced, while the cover of the remaining 13 seeded species that comprised the vigorous legume mix was fairly constant across all tree densities (Fig. 1).

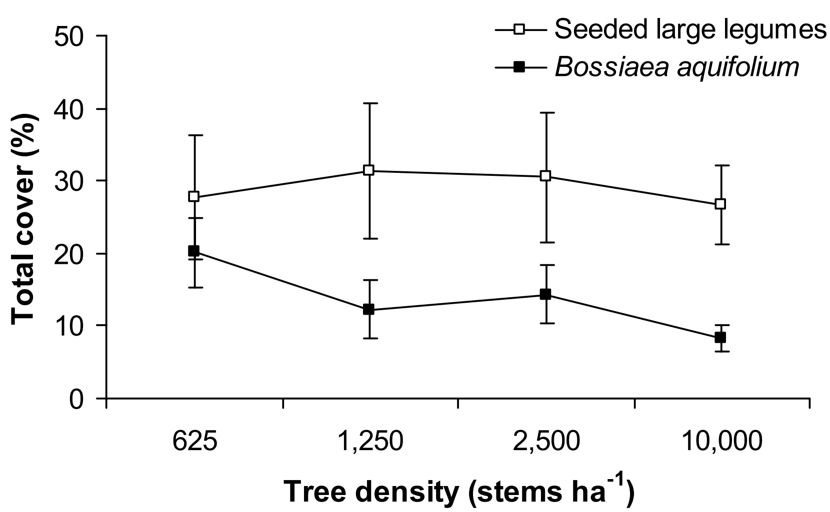

Figure 1. Mean total cover (%) ± S.E. of seeded large legumes (13 species excluding *Bossiaea aquifolium*) and *B. aquifolium* according to tree density (625, 1250, 2500 or 10 000 stems ha^{-1}). Results were non-significant (P > 0.05).

Generally, the effect of the tree density treatment on understorey species richness, plant density and cover was non-significant for most of the seven species groups. Only the species richness of monocots and recalcitrant species were significantly affected (Table 5). Due to the specification of site as a random factor in the ANOVA, pairwise comparisons between tree densities were not possible. However, the species richness of monocots was 5.4 plants/80 m^2 at 10 000 stems ha^{-1} compared with a higher range of 7.6 – 9.0 plants/80 m^2 at the three other tree densities. Species richness of recalcitrant species was 4.4 plants/80 m^2 at 10 000 stems ha^{-1} compared with a higher range of 5.6 - 7.6 plants/80 m^2 at the three other tree densities. Despite the difference in species richness between 10 000 stems ha^{-1} and all other tree densities, there were no consistent trends amongst the lower (625 – 2500 stems ha^{-1}) tree densities.

The effect of the seed treatment on understorey plant density and cover was non-significant for all seven species groups. The species richness of monocots, small non-leguminous shrubs and recalcitrant species were all significantly higher in –L compared with +L seeded sites, while the seed treatment did not affect the species richness of the remaining four groups (Table 5).

Table 5: Species richness (plants/80 m^2), density (plants/m^2) and cover (%) of various species groupings according to tree density (625, 1250, 2500 or 10 000 stems ha^{-1}) and seed treatment (vigorous native legumes [+L] or no vigorous native legumes [-L]). Numbers are means ± S.E. Treatments were analysed in ANOVA using site as a random variable, therefore pairwise comparisons between treatments were not possible. NS = P > 0.05, * = P < 0.05 and ** = P < 0.01. For each tree density value, n = 10 and for each seed treatment value, n = 20.

	Tree density (stems ha^{-1})					Seed treatment		
	625	1250	2500	10 000	Significance	+L	-L	Significance
Species richness/m^2								
Weed species	4.5 ± 0.8	2.9 ± 0.9	4.9 ± 1.2	2.5 ± 0.8	NS	3.4 ± 0.7	4.0 ± 0.6	NS
Monocot species	8.6 ± 1.0	7.6 ± 1.5	9.0 ± 1.3	5.4 ± 0.8	*	6.4 ± 0.9	8.9 ± 0.7	*
Small non-leguminous shrubs	8.7 ± 1.1	6.9 ± 1.7	9.0 ± 1.4	5.9 ± 1.1	NS	6.2 ± 1.0	9.1 ± 0.9	**
Small leguminous shrubs	2.1 ± 0.7	1.9 ± 0.7	1.9 ± 0.5	1.2 ± 0.4	NS	1.3 ± 0.3	2.2 ± 0.5	NS
Recalcitrant species	7.1 ± 0.9	5.6 ± 1.2	7.6 ± 1.5	4.4 ± 0.9	*	5.1 ± 0.8	7.2 ± 0.8	*
Orchid species	4.3 ± 0.6	4.0 ± 0.6	3.5 ± 0.4	2.8 ± 0.4	NS	3.4 ± 0.4	4.0 ± 0.3	NS
Ephemeral species	4.4 ± 0.6	2.6 ± 0.9	5.0 ± 1.4	1.6 ± 0.4	NS	3.0 ± 0.7	3.8 ± 0.7	NS
Plant density/m^2								
Weed species	2.0 ± 0.9	1.0 ± 0.5	2.8 ± 1.0	0.2 ± 0.1	NS	1.1 ± 0.5	1.9 ± 0.6	NS
Monocot species	0.8 ± 0.2	0.8 ± 0.3	0.6 ± 0.1	0.3 ± 0.1	NS	0.6 ± 0.1	0.6 ± 0.2	NS
Small non-leguminous shrubs	1.1 ± 0.3	0.9 ± 0.4	0.8 ± 0.2	0.3 ± 0.1	NS	0.5 ± 0.2	1.0 ± 0.2	NS
Small leguminous shrubs	0.1 ± 0.0	0.2 ± 0.1	0.1 ± 0.0	0.0 ± 0.0	NS	0.0 ± 0.0	0.2 ± 0.1	NS
Recalcitrant species	0.4 ± 0.1	0.5 ± 0.1	0.4 ± 0.1	0.2 ± 0.1	NS	0.3 ± 0.1	0.4 ± 0.1	NS
Orchid species	0.6 ± 0.2	0.4 ± 0.1	0.4 ± 0.1	0.2 ± 0.1	NS	0.5 ± 0.1	0.4 ± 0.1	NS
Ephemeral species	2.2 ± 1.0	2.4 ± 1.9	3.4 ± 1.6	0.1 ± 0.1	NS	1.7 ± 0.9	2.3 ± 1.0	NS
Cover (%)								
Weed species	0.4 ± 0.2	0.3 ± 0.2	0.5 ± 0.2	0.0 ± 0.0	NS	0.2 ± 0.1	0.5 ± 0.2	NS
Monocot species	0.4 ± 0.1	1.3 ± 0.9	0.2 ± 0.0	0.1 ± 0.0	NS	0.4 ± 0.1	0.7 ± 0.5	NS
Small non-leguminous shrubs	6.2 ± 1.6	7.5 ± 2.5	3.8 ± 1.4	1.7 ± 0.4	NS	4.2 ± 1.2	5.3 ± 1.3	NS
Small leguminous shrubs	0.3 ± 0.1	1.1 ± 0.4	0.3 ± 0.2	0.8 ± 0.4	NS	0.5 ± 0.2	0.7 ± 0.3	NS
Recalcitrant species	2.2 ± 0.8	4.0 ± 1.5	0.8 ± 0.2	0.8 ± 0.3	NS	1.7 ± 0.6	2.2 ± 0.7	NS
Orchid species	0.2 ± 0.1	0.2 ± 0.1	0.0 ± 0.0	0.0 ± 0.0	NS	0.2 ± 0.1	0.0 ± 0.0	NS
Ephemeral species	0.3 ± 0.1	0.2 ± 0.1	0.5 ± 0.3	0.0 ± 0.0	NS	0.2 ± 0.1	0.4 ± 0.2	NS

There was a non-significant (P = 0.09) negative linear relationship between total understorey density and tree density (Fig. 2). There was a trend of similar total understorey density at 625 - 2500 stems ha^{-1}, which was reduced at 10 000 stems ha^{-1}.

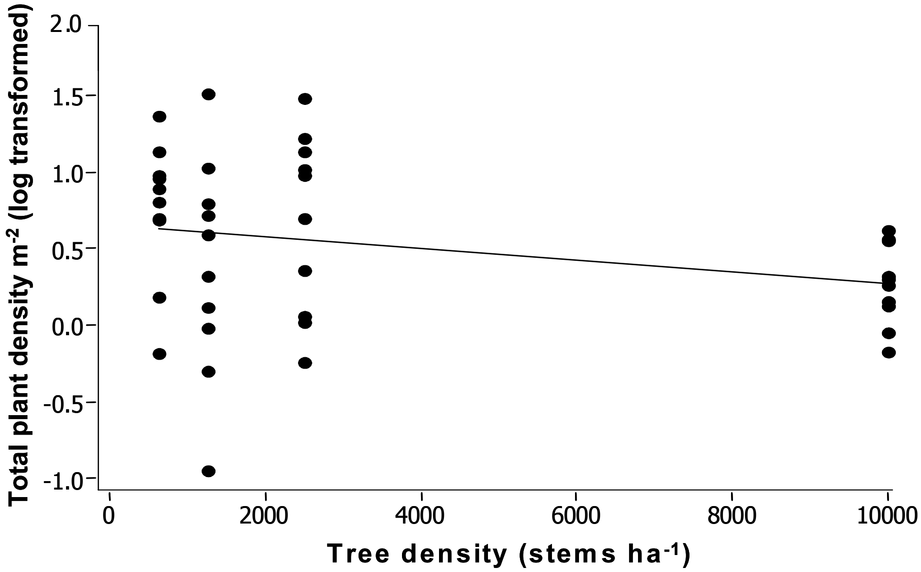

Figure 2. Relationship between total understorey density (plants m^{-2}) and tree density (625, 1250, 2500 or 10 000 stems ha^{-1}). $R^2 = 0.046$, P = 0.09, n = 40. Total plant density was log transformed so that the conditions of normality were satisfied.

Multivariate analyses

Multivariate analysis (ANOSIM) showed that the seed treatment significantly affected total plant cover (P<0.001), site was significant for total plant density and cover, while tree density did not significantly affect either vegetation parameter (Table 6).

Table 6: A summary of the multivariate significant differences between total plant density and cover and site (Karri, Peacock, Possum, Scarp or Wriggly), tree density (625, 1250, 2500 or 10 000 stems ha^{-1}) and seed treatment (vigorous native legumes [+L] or no vigorous native legumes [-L]). NS = P > 0.05, ** = P ≤ 0.01, *** P < 0.001 (n = 40).

Vegetation Parameter	Site	Tree Density	Seed
Total plant density/m^2	***	NS	NS
Total plant cover	**	NS	***

Multivariate analysis showed that the effect of tree density on understorey density was non-significant (P = 0.085), therefore pairwise comparisons were not performed (Fig. 3). However, there was separation of the 10 000 and 625 stems ha^{-1} treatments on the

MDS, while the intermediate tree densities were dispersed throughout the ordination hyper-space.

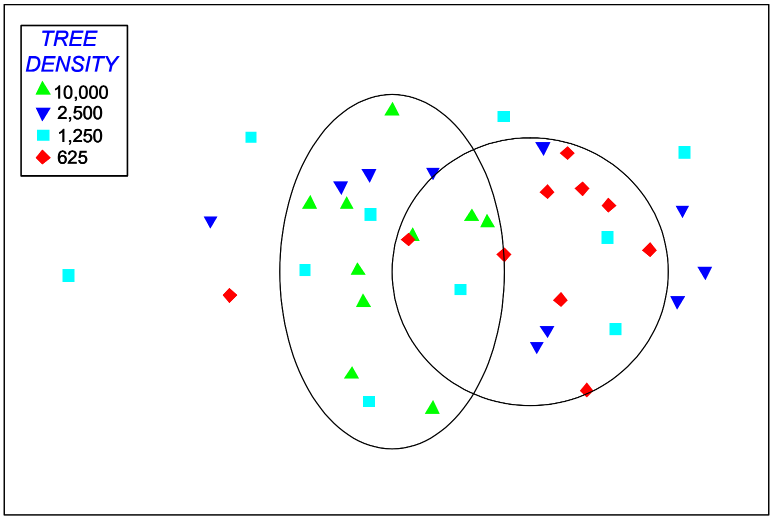

Figure 3. MDS ordination of the 40 sample plots based on understorey density in relation to tree density (625, 1250, 2500 or 10 000 stems ha^{-1}). R = 0.09, P = 0.085, stress = 0.17. The superimposed circles display the separation of 625 (♦) and 10,000 (▲) stems ha^{-1}. Results were non-significant (P > 0.05).

Multivariate analysis showed that the effect of tree density on understorey cover was non-significant (P = 0.067), therefore pairwise comparisons were not performed (Fig. 4). The 10 000 stems ha^{-1} sites cluster in the middle of the ordination hyper-space, while the remaining tree densities were dispersed.

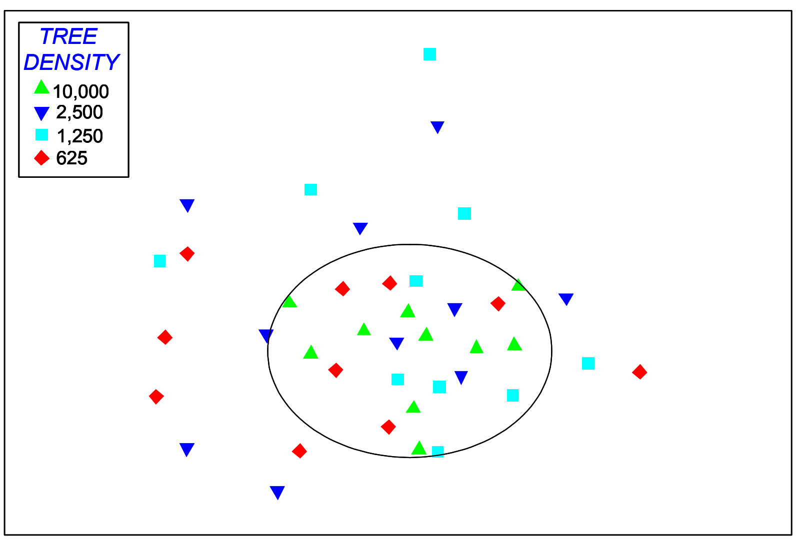

Figure 4. MDS ordination of the 40 sample plots based on understorey cover in relation to tree density (625, 1250, 2500 or 10 000 stems ha^{-1}). R = 0.09, P = 0.067, stress = 0.21. The superimposed circle displays the grouping of the 10 000 (▲) stems ha^{-1} sites. Results were non-significant (P > 0.05).

Multivariate analysis showed the effect of the seed mix on understorey cover was significant (P < 0.001, Fig. 5). There was separation of sites seeded with and without the vigorous native legumes in the MDS ordination hyper-space.

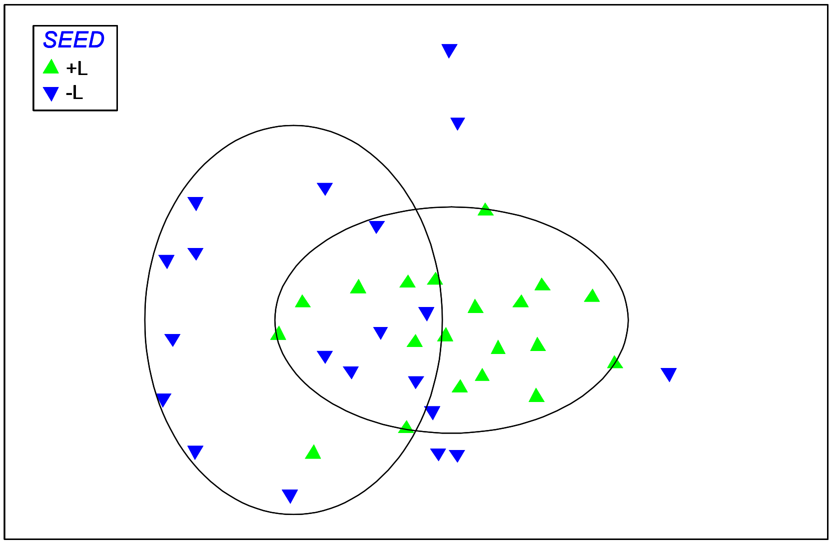

Figure 5. MDS ordination of the 40 sample plots based on species cover (%) in relation to seed treatment. R = 0.25, P < 0.001, stress = 0.21. The superimposed circles display the separation of the vigorous native legume seed (+L ▲) and no vigorous legume seed (-L ▼).

DISCUSSION

Tree Density

Generally, the effect of tree density was non-significant for the majority of the understorey vegetation parameters, individual species, genera and species groups of interest. The exception was the species richness of the monocot and recalcitrant species groups, which were significantly affected by tree density. Although a statistical comparison of the species richness at each tree density was not possible due to the analysis employed, there was a trend of higher species richness at the three lower tree densities compared to 10 000 stems ha^{-1}. Monocots are a highly desirable species group in rehabilitated sites. Over 90% of the monocots present in the current study were resprouting species, while the remaining monocots were grasses and other seeder species. Resprouting species have the ability to regenerate from buds or below ground tissues such as lignotubers, rhizomes, bulbs or corms (Bell *et al.* 1984) and are able to recover from natural disturbances such as fire, drought and insect attack (Bellairs and Bell 1993). Over 70% of species in the unmined jarrah forest are resprouters (Bell and Koch 1980), while rehabilitated areas are dominated by nonresprouting seeder species (Grant and Loneragan 1999). Increasing the abundance of resprouting species results in increased resilience to disturbance and a species composition more similar to the jarrah forest (Bellairs and Bell 1993). The restored vegetation is also visually more similar to the unmined jarrah forest due to the establishment of a 'grass-like' vegetation component. Similarly, the increased species richness of recalcitrant species at lower tree densities is highly desirable, because these species are inherently difficult to restore in rehabilitated sites as they have very poor seed set and propagate vegetatively. Approximately 100 000 tissue cultured plants of 12 recalcitrant species are planted in rehabilitated sites annually (Willyams 2005), a program that comes at a substantial cost to Alcoa's rehabilitation operations (Koch 2007, *in press*).

For three of the understorey vegetation parameters (total species richness, density and cover), the effect of tree density was nearly significant ($0.05 < P < 0.10$). There was significant variation in the understorey vegetation between experimental sites, which may have masked some of the differences between the tree density treatments. In addition, an indirect effect of tree density is that the vigorous large legumes may grow better at wider tree spacings and out-compete the smaller understorey plants. Therefore, the higher richness, density and cover of understorey species that might be expected in a more open tree canopy are negated by the compensatory growth of the large legumes. Large legumes can originate both from the applied seed mix and natural seedbank in the returned topsoil. The significantly higher density and cover of *Bossiaea aquifolium* in sites with no vigorous legume seed applied shows it established well from the natural seed bank. In addition, while there was a reduction in the species richness of monocot and recalcitrant species at high tree densities (10 000 stems ha^{-1}), there were few differences among the three lower tree densities (range of 625 – 2500 stems ha^{-1}). The large difference in tree stocking between 10 000 stems ha^{-1} and the other tree spacings may explain why lower understorey richness was only observed at this tree density.

High eucalypt density can result in tree mortality and slow growth rates (Koch and Ward 2005) and increased water use (Croton 2004). Unlike many other *Eucalypt* species, jarrah does not undergo self-thinning at high stand densities (Abbott and Loneragan 1986). In another study at three of the same sites (C. McFarlane *et al.*

unpublished data), mean projected foliage cover of the canopy measured 18 years post-establishment (–L seed, 500 kg ha^{-1} superphosphate) was 63, 53, 43 and 24% at 10 000, 2500, 1250 and 625 stems ha^{-1} respectively. Sites with high tree densities may compete with the understorey vegetation for two reasons; 1) the direct effect of shading from higher canopy cover and 2) the indirect effect of higher competition for water. In natural *Pinus sylvestris* forest in the eastern Pyrenees, forest structural variables such as tree basal area or tree density were not important in determining the diversity of understorey species (Pausas 1994). Rather, moisture-related parameters and soil nutrient concentrations were the main factors in predicting the species richness of the understorey of this forest type (Pausas 1994). In naturally occurring *Eucalyptus* communities in central Queensland, the understorey herbaceous yield decreased as tree basal area increased, however the level of reduction was related to the productivity of the site (Scanlan and Burrows 1990).

Alcoa currently aims for 1300 stems ha^{-1} initial establishment rate, which approximates the 2 m x 4 m tree spacing (1250 stems ha^{-1}). Target tree densities were reduced in 2000 from 2500 to 1300 stems ha^{-1} due to concerns that high tree densities may compromise other forest values (Grant 2006). In 2004, an upper tree density limit of 2500 stems ha^{-1} was introduced. Rehabilitated sites with tree densities above this limit are spot sprayed with herbicide at nine months of age. Tree densities within the acceptable range (600 – 2500 stems ha^{-1}) will not significantly impact the richness, density or cover of understorey established in rehabilitated sites.

For rehabilitated sites that are densely stocked prior to the introduction of the upper density limit, trees can be thinned to 1,111 (16 m^2 ha^{-1}, 3 x 3 m spacing) and 625 stems ha^{-1} (11 m^2 ha^{-1}, 4 x 4 m spacing) at 10-13 years of age without any significant impact on the native understorey vegetation (Grant and Norman 2006). Although there was no change in native species richness, evenness or diversity of the understorey when measured 18 months post-thinning, over time, understorey recruitment may increase due to more light and less inter-species competition for water and nutrients. However, Neyland and LaSala (2005) noted no change in understorey species composition, cover and abundance four to five years after thinning of even-aged eucalypt regeneration stands (*Eucalyptus obliqua* and *E. regnans*) in southern Tasmania from 23.1-36.1 m^2 ha^{-1} down to a similar basal area of

8.0-12.7 m^2 ha^{-1}. In contrast, positive results to thinning such as increased understorey species richness and cover have been noted after a longer time-frame (12-16 years post-thinning) in even-aged Douglas-fir stands in the USA (Thomas *et al.* 1999).

In general, the species richness of the rehabilitated sites was similar to that previously reported for similar aged rehabilitated sites. Species richness ranged from 22.1 - 34.8 species/80 m^2, similar to the average of 32.5 species/80 m^2 recorded in 12- to 15-year-old rehabilitated sites (Grant and Norman 2006), but lower than the 41.5 species/80 m^2 recorded in 14-year-old rehabilitated sites (Norman *et al*. 2006).

Seed Mix

The suppressant effect of a dense legume understorey on plant density and the number of smaller shrub and herb species noted by Koch and Davies (1993) nine to 12 years after seeding was evident in 15-year-old rehabilitated sites. Initially, the +L seed mix produced significantly higher species richness, density and cover than the –L seed mix when monitored 9 months after seeding (Koch and Ward 1994). At 15 years of age, there was still higher plant cover in +L seeded sites, while native species richness was higher in -L seeded sites and plant density was not significantly affected by the applied seed mix. The density of six and cover of four small native understorey species were all significantly higher in –L compared with +L seeded sites. Three of these species are recalcitrant (*Boronia fastigiata*, *Lasiopetalum floribundum* and *Lomandra sonderi*), therefore their increased representation in rehabilitated sites is desirable. Four genera (excluding genera that contained significant species from the individual species analysis) were also significantly higher in –L compared with +L seeded sites. The species richness of monocots, small non-leguminous shrubs and recalcitrant species was also significantly higher in –L compared with +L seeded sites. Koch and Davies (1993) also found that sparse *Acacia* areas had higher species richness, density, cover and diversity (an increase of 53%, 41%, 31% and 84% respectively) of small shrub and herb species compared with the dense acacia areas.

The higher density of the *Caladenia* genus (Orchidaceae) in –L seeded sites is of interest, because orchid seed is not added to seed mixes as it is minute and readily dispersed by wind (Rasmussen 1995). Establishment of orchids in rehabilitated areas requires the recovery of the inoculum potential of mycorrhizal fungi and establishment of suitable microhabitats through the regrowth of vegetation (Grant and Koch 2003). However, when orchids of different genera were grouped for analysis, there was no effect of seed treatment on density or cover. The recovery of orchids in relation to vegetation structure and species diversity is of interest to identify suitable orchid habitats in rehabilitated sites and is being investigated (Collins 2005).

While the acacias begin to senesce after eight years, it appears the smaller species do not invade +L seeded sites after 15 years of growth. Koch and Davies (1993) suggested a longer period of time or a disturbance may be required to facilitate recruitment. Norman *et al*. (2006), found there was no difference in native species richness measured 2, 5, 8 and 14 years post-rehabilitation seeded with the +L and -L seed treatments. The acacias and other tall legumes may have displaced the small species, resulting in similar total plant species richness despite a reduction in the number of smaller herbs and shrubs.

Rehabilitation efforts of mining companies have historically focused on establishing rapid-growing species that control erosion but may inhibit the establishment of smaller native species. In the eastern USA, some of the aggressive legumes initially established on the rehabilitated coal mines to provide vegetative cover may have slowed long-term vegetation recovery, however site differences and changes in rehabilitation practices over time complicated the results (Holl 2002). Chapman *et al.* (1996) also noted that aggressive growth by legumes may restrict the diversity of species-rich meadows re-created on sites restored after coal mining in the United Kingdom. The highly competitive legumes and grasses established following phosphate mining in Idaho, USA, have also resulted in limited establishment of native species (Chambers *et al.* 1994). It was suggested that the establishment of native species could be increased and natural successional processes facilitated by omitting the competitive legumes and including less-competitive native legumes (Chambers *et al.* 1994). The understorey seed mix used in 2005 mine rehabilitation had a much reduced quantity of large legumes (0.4 kg ha^{-1}) compared with the 1988 seed mix (2 kg ha^{-1}). It is believed this will allow better establishment and survival of smaller understorey species while still providing the benefits associated with these native legumes.

CONCLUSIONS

The applied seed mix had a significant effect on the understorey vegetation in 15-year-old rehabilitated sites, while the effect of tree density was less pronounced . The removal of large vigorous native legumes from the seed mix significantly increased total species richness, the density and cover of a number of individual species and genera, and the species richness of monocots, small non-leguminous shrubs and recalcitrant species. High tree density (10 000 stems ha^{-1}) significantly reduced the species richness of monocots and recalcitrant species, and the cover of one individual species (*Opercularia echinocephala*). The general lack of response to tree density could be due to the large variation in understorey between sites and the compensatory effect of the large legumes, in particular *Bossiaea aquifolium*, at lower tree densities. Generally, there were few differences in the understorey vegetation between 625 – 2500 stems ha^{-1} compared with 10 000 stems ha^{-1}. The target tree density for rehabilitated sites is 1300 stems ha^{-1} (approximately 2 m x 4 m spacing). The rate of vigorous legume seed has been reduced to less than 0.4 kg ha^{-1} based on the results of this and other research.

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REFERENCES

Abbott, I. and Loneragan, O. W. (1986). *Ecology of jarrah (*Eucalyptus marginata*) in the northern jarrah forest of Western Australia.* CALM Bulletin No 1. The Department of Conservation and Land Management: Perth.

Bell, D. T., Hopkins, A. J. M. and Pate, J. S. (1984). Fire in the kwongan. In '*Kwongan: plant life of the sandplain*.' (Eds. J. S. Pate and J. S. Beard). pp. 178-204. University of Western Australia Press: Nedlands.

Bell, D. T. and Koch, J. M. (1980). Post-fire succession in the northern jarrah forest of Western Australia. *Australian Journal of Ecology* **5**: 9-14.

Bellairs, S. M. and Bell, D. T. (1993). Seed stores for restoration of species-rich shrubland vegetation following mining in Western Australia. *Restoration Ecology* **1**: 231-240.

Chambers, J. C., Brown, R. W. and Williams, B. D. (1994). An evaluation of reclamation success on Idaho's phosphate mines. *Restoration Ecology* **2**: 4-16.

Chapman, R., Collins, J. and Younger, A. (1996). Control of legumes in a species-rich meadow re-created on land restored after opencast coal mining. *Restoration Ecology* **4**: 407-411.

Collins, M., Koch, J., Brundrett, M. and Sivasithamparam, K. (2005). Recovery of terrestrial orchids in the post-mining landscape. *Selbyana* **26**(1,2): 255-264.

Conservation Commission of Western Australia. (2004). Forest Management Plan 2004-2013. The Conservation Commission of Western Australia: Perth. pp. 11.

Croton, J. T. (2004). *Estimating the hydrologic response of the Del Park catchment to bauxite mining.* Environmental Research Bulletin No. 32. Alcoa World Alumina Australia: Perth.

Department of Industry and Resources (DoIR) (2002). *Alcoa World Alumina Australia Darling Range bauxite mine rehabilitation completion criteria*. DoIR: Perth.

Elliott P, Gardner J, Allen D, Butcher G (1996) Completion Criteria for Alcoa of Australia Limited's Bauxite Mine Operation. In '*Proceedings of the 3rd International 21st Annual Minerals Council of Australia Environmental Workshop*'. pp. 79-89. Minerals Council of Australia: Canberra.

Florabase (2006). http://www.calm.wa.gov.au/florabase/index.html

Gardner, J. (2001). Rehabilitating bauxite mines to meet land use objectives: bauxite mining in the jarrah forest of Western Australia. *Unasylva* **52**: 3-8.

Grant, C. D. (2006). State-and-transition successional model for bauxite mining rehabilitation in the jarrah forest of Western Australia. *Restoration Ecology* **14**: 28-37.

Grant, C. D. and Koch, J. (2003). Orchid species succession in rehabilitated bauxite mines in Western Australia. *Australian Journal of Botany* **51**: 453-457.

Grant, C. D. and Loneragan, W. A. (1999). The effects of burning on the understorey composition of 11-13 year-old rehabilitated bauxite mines in Western Australia. *Plant Ecology* **145**: 291-305.

Grant, C. D. and Norman, M. A. (2006). *Understorey response to thinning and burning operations in 10- to 13-year-old rehabilitated bauxite mines in the* *jarrah forest.* Environmental Department Research Bulletin No. 35. Alcoa World Alumina Australia: Perth.

Holl, K. D. (2002). Long-term vegetation recovery on reclaimed coal surface mines in the eastern USA. *Journal of Applied Ecology* **39**: 960-970.

Koch, J. M. (2007). Restoring a jarrah forest understorey vegetation after bauxite mining in Western Australia. *Restoration Ecology, in press.*

Koch, J. and Davies, S. (1993). *The effect of a tall dense Acacia understorey on small shrub and herb species native to the jarrah forest*. Environmental Department Research Bulletin No. 21. Alcoa of Australian Limited: Perth.

Koch, J. M. and Ward, S. C. (1994). Establishment of understorey vegetation for rehabilitation of bauxite-mined areas in the jarrah forest of Western Australia. *Journal of Environmental Management* **41**: 1-15.

Koch, J. M. and Ward, S. C. (2005). Thirteen year growth of jarrah (*Eucalyptus marginata*) on rehabilitated bauxite mines in south-western Australia. *Australian Forestry* **68**: 176-185.

Neyland, M. G. and LaSala, A. V. (2005). Response of understorey floristics to pre-commercial thinning and fertilising in even-aged eucalypt regeneration. *Tasforests* **16**; 71-82.

Norman, M. A., Koch, J. M., Grant, C. D., Morald, T. K. and Ward, S. C. (2006). Vegetation succession after bauxite mining in Western Australia. *Restoration Ecology* **14**: 278-288.

Pausas, J. G. (1994). Species richness patterns in the understorey of Pyrenean *Pinus sylvestris* forest. *Journal of Vegetation Science* **5**: 517-524.

Rasmussen, H. N. (1995). Properties of 'dust' seeds. In '*Terrestrial Orchids from Seed to Mycotrophic Plant.*' Cambridge University Press: UK. pp. 7-16.

Scanlan, J. C. and Burrows, W. H. (1990). Woody overstorey impact on herbaceous understorey in *Eucalyptus* spp. communities in central Queensland. *Australian Journal of Ecology* **15**: 191-197.

Thomas, S. C., Halpern, C. B., Falk, D. A., Liguori, D. A. and Austin, K. A. (1999). Plant diversity in managed forests: understory responses to thinning and fertilization. *Ecological Applications* **9**: 864-879.

Willyams, D. (2005). Tissue culture of geophytic rush and sedge species for revegetation of bauxite mine sites in the northern jarrah forest of Western Australia. pp. 226-241. In '*Contributing to a Sustainable Future*'. (Eds. I. J.

Bennett, E. Bunn, H. Clarke and J. A. McComb). Proceedings of the Australian Branch of the International Association for Plant Tissue Culture and Biotechnology: Perth.