Measuring Success: Evaluating the Restoration of a Grassy Eucalypt Woodland on the Cumberland Plain, Sydney, Australia

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Abstract

We compared the floristic composition and structure of restoration areas of eucalypt woodland with untreated pasture (control) and remnant vegetation (reference) in western Sydney. The restored areas comprised over 1,000 ha of abandoned pasture, which had been treated to reduce weeds and planted with seedlings of 26 native plant species raised from seed obtained locally from remnant vegetation. Plantings were carried out 0–9 years ago. Floristic composition was measured in quadrats using frequency scores and cover abundance. As far as possible treatments and restoration ages were replicated across sites. Ordination and analyses of similarity failed to distinguish the composition of restored vegetation from that of untreated pasture, which were both significantly different from that of remnant vegetation. There was a weak compositional trend with age of restored vegetation, but this was not in the direction of increasing resemblance to remnant vegetation. There was some evidence for convergence in structural features of restored with remnant vegetation, but this was at least partly attributed to plant growth. Subject to constraints imposed by the sampling design, environmental factors, and spatial variation were discounted as explanations for the results. The results therefore suggest either failure of restoration treatments or a restoration trajectory that is too slow to detect within 10 years of establishment. Our conclusions agree with those of similar studies in other ecosystems and support: (1) the need to monitor restoration projects against ecological criteria with rigorous sampling designs and analytical methods, (2) further development of restoration methods, and (3) regulatory approaches that seek to prevent damage to ecosystems rather than those predicated on replacing losses with reconstructed ecosystems.

Key words: ecological audit, field experiment, mitigation policy, restoration trajectory, succession.

Introduction

Restoration of native ecosystems is now a widely recognized imperative for both nature conservation and sustainable production (WRI, IUCN & UNEP 1992; Hobbs 1993). For example, in 2000–2001 the Australian government spent $36.4 million of its Bushcare Program on community grants for "practical works to re-establish native vegetation to provide habitat for wildlife, rehabilitate degraded lands and protect native vegetation" (Environment Australia 2001). Projects carried out under this and other programs enjoy extensive support in rural and urban communities, which more than doubles the monetary input with in-kind contributions of volunteer labor, knowledge capital, and equipment. The total financial input is likely to be substantially higher when expenditure by state governments and industry are accounted for. The success of these projects is reported in terms of administrative indicators. For example, Environment Australia (1999) reported the "major on-ground outputs" of its $27.1 million Bushcare funding included 10,000 ha directly revegetated, 4.5 million tubestock planted, and 12,000 km of fencing constructed. Yet these measures tell us little about the biological outcomes of the projects that are funded and implemented.

Unless administrative audits of the type routinely published in the annual reports of funding agencies are accompanied by ecological audits, it is impossible to know how much the sums invested, areas treated, and community commitment actually contributed to restoration of degraded ecological assets. A meaningful ecological audit of restoration projects must address the extent to which the restored areas follow a trajectory toward some specified target state that represents "natural" or undegraded conditions (Hobbs & Mooney 1993; Hobbs & Norton 1996; Zedler & Callaway 1999). Success may be assessed by measuring aspects of species composition, community structure, and ecosystem function.

Chapman and Underwood (2000) recently drew attention to the need for a scientific protocol to measure the biological success of restoration. A central component of their proposition was comparative monitoring and experimentation to address hypotheses about whether restored sites increase their resemblance to reference sites that represent a target state. Such an approach poses challenges to ecologists in deciding how to choose reference sites, how to select response variables and measure resemblance, and

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how to design sampling in a way that minimizes influence of confounding factors. In this article we attempt to address these challenges by assessing an extensive 10-year project near Sydney, Australia that aims to restore an endangered ecological community by planting on agricultural land. In this study we applied a replicated comparative sampling design to determine whether the composition and structure of restoration plantings are on a trajectory from abandoned exotic pasture toward comparatively undisturbed remnant vegetation. We do not attempt to measure ecosystem functions directly, as some other studies have (e.g., Zedler & Callaway 1999), but seek to draw inferences about functions from compositional and structural data (Reay & Norton 1999).

Methods

Study Sites

The Cumberland Plain (34°S, 151°E) is a shale valley in a coastal rainshadow surrounded by extensive sandstone plateaux. The plain contains the western portion of Sydney, Australia's largest city. The predominantly clay soils of the plain supported distinctive grassy eucalypt woodlands, which have been reduced to 9% of their original extent by extensive clearing for agriculture and subsequent urban development (Benson & Howell 1990). Remnant patches are small, and more than two-thirds of their remaining area is degraded by eutrophication, weed invasion, and human disturbances. The woodlands are now recognized as an endangered ecological community under state and national legislation (NSW Threatened Species Conservation Act 1995, Commonwealth Environment Protection and Biodiversity Conservation Act 1999).

The study sites were located on farmland that was acquired by state government agencies as reserve or catchment land. They were selected because they included restored vegetation of varying age juxtaposed with patches of remnant vegetation and abandoned pasture. Based on nearby remnants and habitat models (Tozer 2000), we inferred that all sites were originally woodland dominated by trees of the Myrtaceae, *Corymbia maculata* (spotted gum), *Eucalyptus moluccana* (grey box), *E. crebra* (narrow-leaved ironbark), and *E. tereticornis* (forest red gum) with an understory including a patchy shrub stratum of *Bursaria spinosa* (Pittosporaceae), *Dillwynia sieberi* (Fabaceae), and *Indigofera australis* (Fabaceae) and a semi-continuous ground cover of grasses such as *Themeda australis*, *Echinopogon caespitosus*, and *Entolasia marginata* (all Poaceae) and herbs including *Brunoniella australis* (Acanthaceae), *Dichondra repens* (Convolvulaceae), and *Pratia purpurascens* (Lobeliaceae) (Benson 1992). Nomenclature follows Harden (1990–2002).

Restoration projects commenced in the area in 1992 and have been managed by a single authority (Greening Australia) since then. Historically, the sites were grazed by cattle, fertilized, and sown to exotic pasture grasses, particularly *Phalaris* spp. (Poaceae). The stated goals of the restoration project include the "re-establishment of native vegetation" (Perkins 1997). The restoration plantings were carried out in a pattern designed to connect remnant patches of woodland, which were also the primary sources of seed for tubestock. To evaluate success against the above goal we therefore identified the remnants as suitable reference sites to which the restored sites were expected to increase their resemblance in composition and structure over time. The untreated abandoned pasture was defined as a control from which restored sites were expected to become increasingly dissimilar in composition and structure with time.

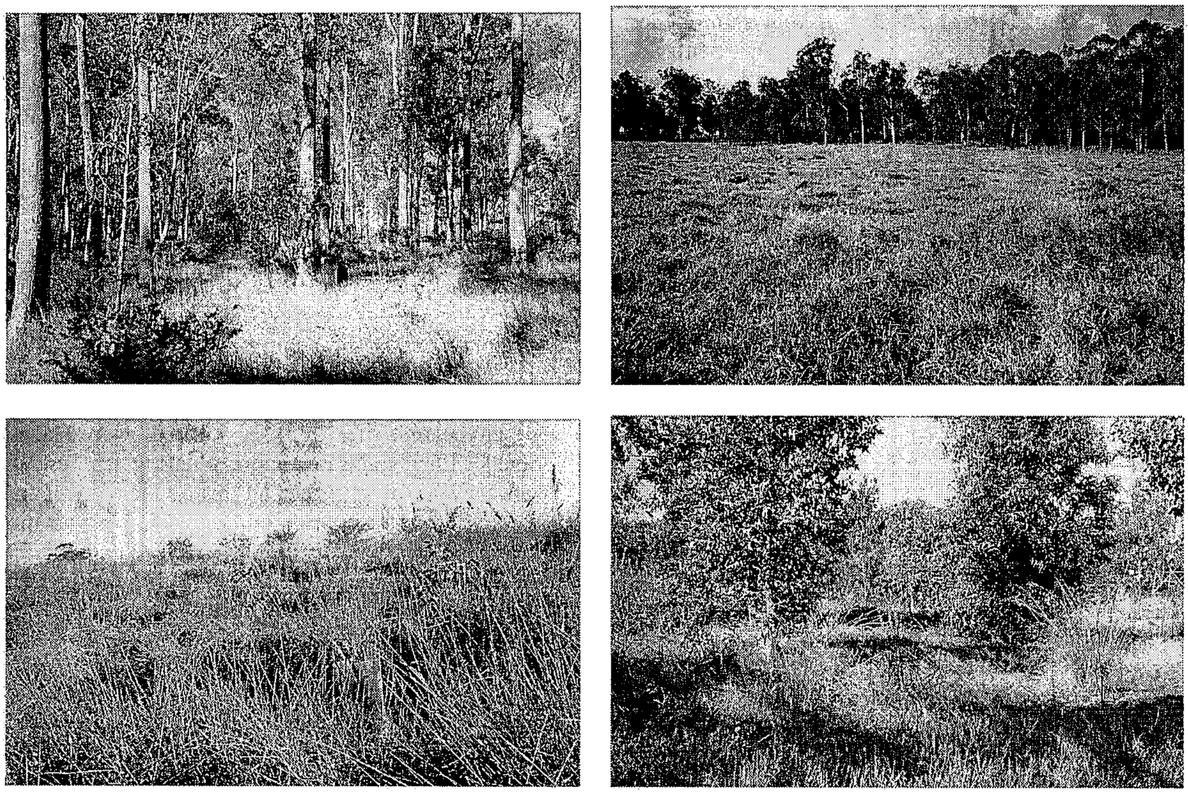
The restoration process was initiated with weed control (J. Christie 2001, Greening Australia, personal communication) at the first sites in 1992. All sites were slashed and sprayed with glyphosate before planting was undertaken. Twenty-six indigenous tree and shrub species, propagated to tubestock from local seed sources, were planted mechanically in rows after the pasture began to break down. The mix of planted species varied across the landscape, the aim being to match species with soils and topographic positions occupied by their wild populations. All plants were weed matted with a recycled paper disk and surrounded by a protective plastic sleeve. Maintenance sprays of glyphosate were applied in spring and autumn for 2 to 3 years after planting to reduce competition from weeds in the vicinity of plants. To reduce the risk of fire mechanical slashing was carried out among the plantings and hazard reduction fires were lit in areas surrounding plantings at approximately annual intervals. Since 1998 fencing was constructed to exclude livestock from restoration areas and remnants (D. Williams 2001, Greening Australia, personal communication).

Selection of Sampling Sites

Sites in close proximity and with similar topography (upper and mid-slopes) were selected to minimize environmental variation that might potentially confound management effects. It was possible to sample three different management treatments (untreated pasture, restored vegetation, and remnant vegetation) and four different ages of restored vegetation across four sites (Table 1). However, a fully orthogonal

Table 1. Number of samples in each combination of management treatment and site.

Management Treatment	*Hoxton Park*	*Plough and Harrow Property*	*Western Sydney Regional Park*	*Prospect Reservoir*
Untreated pasture	3	3	0	0
1-year-old restoration	3	0	0	0
3-year-old restoration	3	0	0	0
6-year-old restoration	2	2	0	0
9-year-old restoration	0	0	3	0
Remnant vegetation	3	0	0	3



A B C D

Figure 1. Examples of management treatments: (A) remnant vegetation, (B) untreated pasture, (C) restored vegetation 1 year after establishment, and (D) restored vegetation 6 years after establishment.

sampling design was not available. Examples of these treatments are shown in Figure 1. It is acknowledged that disturbance resulting from past agricultural practices in the area have impacted on remnant patches to varying degrees, but we assumed that these effects were randomized across sites.

Data Collection

Three samples were placed randomly in each available combination of management treatment, restoration age, and site, except for the 6-year-old restoration treatment, which could only be sampled twice due to the spatial design of plantings (Table 1). Vascular plant species composition was recorded using the frequency score method (Morrison et al. 1995) and Braun-Blanquet cover abundance estimates. A frequency score was computed for each species by counting the number of occurrences out of six subquadrats. The subquadrats were in a nested square layout in which the dimensions were successively doubled in a geometric sequence (1, 2, 4, 8, 16, and 32 m). These scores provide a sensitive measure of abundance that is correlated with plant density (Morrison et al. 1995). Differences in frequency scores between pasture and restoration treatment were therefore expected to indicate recruitment of new individuals rather than changes in cover alone. Braun-Blanquet cover-abundance estimates were recorded within a subquadrat of 20 × 20 m (1, <5% cover and one or a few individuals; 2, <5% cover and uncommon; 3, <5% and common; 4, <5% and very abundant or 5–20% cover; 5, 20–50% cover; 6, 50–75% cover; and 7, >75% cover). There was a substantial volume of such cover-abundance data from a previous survey of remnant vegetation over a larger area of the Cumberland Plain. We therefore recorded data in the same format to compare our pasture and restored sites with remnant vegetation over a broader geographic domain than that permitted by our own samples. Planted and wild occurrences of the same species were recorded separately.

Average height and cover of each vegetation stratum were visually estimated to assess vegetation structure. Percentage cover of bare ground and leaf litter (estimated visually) and environmental covariables, including aspect, slope, soil texture, and grid location, were also recorded.

Data Analysis

Species Composition. Bray-Curtis dissimilarity matrices were computed from the frequency score data for (1) all native and exotic vascular plants (including both wild and planted occurrences) and (2) wild occurrences of native and exotic vascular plants only (planted occurrences excluded). All analyses were applied to both data sets. Semistrong hybrid multidimensional scaling ordinations (Belbin 1991) were calculated in two and three dimensions to obtain a graphical representation of floristic relationships between management treatments. Stress values, which indicate the degree to which distances between samples in ordination space resemble dissimilarity values, were used to evaluate the goodness of fit for the two- and three-dimension ordinations. Linear vectors representing the target trajectory and the observed trajectory of revegetated sites were fitted indirectly to the ordination using principal axis correlation, a form of multiple linear regression (Belbin 1994). The "target" vector represented a direct transition from pasture (value = 0) to remnant vegetation (value = 1), with all revegetation samples coded as missing values. The "observed" vector represented revegetation age (taking values of 1, 3, 6, and 9 years, respectively), with remnant and pasture samples coded as missing values. Developmental trends were also represented by a line joining the centroids of each age class.

To examine the revegetation samples in the context of a larger and more spatially extensive sample of remnant vegetation, a second data matrix was constructed by combining the cover-abundance data collected in this study (25 samples, planted individuals excluded) with a further 33 samples gathered from woodland remnants within a 5-km radius of our study area. These latter data were from a regional vegetation survey of the remnant Cumberland Plain woodlands (Tozer 2000), which used an identical survey protocol to that described here for cover-abundance estimates. The additional samples were all from mid- and upper slopes on Wianamatta shale to ensure similar sampling constraints as imposed in our design. An ordination was calculated in two and three dimensions from a Bray-Curtis dissimilarity matrix, as described above, except that dissimilarities were calculated from cover-abundance estimates rather than frequency scores.

Analyses of similarity (ANOSIMs) (Clarke & Gorley 2001) were used to test for differences in species composition between management treatments (pasture, restored, and remnant) and restoration ages (cleared, 1 year, 3 years, 6 years, 9 years, and remnant). SIMPER analyses (Clarke & Gorley 2001) were carried out to determine the contribution of each species to the average sample dissimilarity between significantly different management treatments.

In the ordinations and analyses of similarities described above, site differences potentially confounded differences between management treatments because it was necessary to pool data from different sites to obtain a full comparison of management treatments and revegetation ages. A two-way ANOSIM was therefore carried out on an orthogonal portion of the data (pasture and 6-year-old revegetation at Hoxton Park and Plough and Harrow; Table 1) to test simultaneously for differences between those management treatments and sites. Differences between sites were also examined in remnant vegetation (Hoxton and Prospect) using a one-way ANOSIM.

To further examine factors that potentially confound differences between management treatments, components of variation attributable to environmental, spatial, and management indicators were quantified using canonical correspondence analysis (ter Braak & Smilauer 1998). Data matrices were constructed for each of the three sources of variation. The environmental matrix included slope, position on slope (upper, middle, or lower), aspect index (a sine-transformation of half the aspect value in degrees; Keith & Bedward 1999), angular elevation to the northern horizon, and altitude. The spatial matrix included x and y grid coordinates and their derivatives x^2, y^2, xy, x^2y, y^2x (Legendre & Legendre 1998) and site (Hoxton Park, Plough and Harrow, Western Sydney Regional Park, and Prospect Reservoir). The management matrix included age and management treatment (pasture, restored, remnant). Table 2 describes the step-wise analytical protocol used to partition variation in each vegetation matrix among environmental, spatial, and management indicators and to all pair-wise and three-way combinations of these sources (after Henderson & Keith 2002). All combinations were tested using 199 Monte Carlo permutations (ter Braak & Smilauer 1998).

Differences in Univariate Community Properties. One-factor analyses of variance and linear models were applied to test for differences between management treatments in vegetation structure and native/exotic composition. Tukey's multiple comparisons tests were used to examine significant results. The following variables were tested: number of exotic species, proportion of exotic species, number of native species ("all species" and "nonplanted individuals only"), percent ground cover, canopy height, and canopy crown cover. Linear models with Poisson error distributions were used to test the numbers of exotic and native species, whereas the other variables were tested using analyses of variance with transformation as required.

To examine the influence of site variation a fixed two-factor analysis of variance was carried out on the orthogonal subset of data described previously for ANOSIMs, and t-tests were carried out on the data from remnant sites at Hoxton and Prospect. Data were log-transformed where Bartlett's test indicated heterogeneous variances. Where variances could not be homogenized, a reduced critical p value was used to assess significance (Underwood 1997).

Linear models (normal and Poisson, as described previously) were fitted to test relationships between restoration age and univariate community properties. All analyses were carried out on the full data set and repeated with planted occurrences excluded.

Table 2. Stepwise analytical protocol and equations for partitioning floristic variation among management treatments, environmental variables, and space using canonical correspondence analysis (CCA).

Step	*Description of Step*	*Equation*
1	Unconstrained CA on vegetation matrix	A = m + e + s + me + ms + es + mes + U
2	CCA on vegetation matrix constrained by disturbance matrix	B = m + me + ms + mes
3	CCA on vegetation matrix constrained by disturbance matrix with combined environment + space matrix as covariable	C = m
4	CCA on vegetation matrix constrained by disturbance matrix with environment matrix only as covariable	D = m + ms
5	CCA on vegetation matrix constrained by disturbance matrix with space matrix only as covariable	E = m + me
6	CCA on vegetation matrix constrained by environment matrix with combined disturbance + space matrix as covariable	F = e
7	CCA on vegetation matrix constrained by environment matrix with disturbance matrix only as covariable	G = e + es
8	Sum of eigenvalues for vegetation matrix constrained by space matrix with combined disturbance + environment matrix as covariable	H = s
9	Calculate disturbance and environment overlap	me = E − C
10	Calculate disturbance and space overlap	ms = D − C
11	Calculate environment and space overlap	es = G − F
12	Calculate disturbance, environment and space overlap	mes = B− (D) − (E − C)
13	Calculate unexplained variation	U = A − (B) − (F) − (H) − (G-F)
14	Check additivity of components with other permutations of formulae	

Sums of eigenvalues for various CCAs are given by constants A–H. Components of variation in vegetation matrix are as follows: m, uniquely attributable to management; e, uniquely attributable to environment; s, uniquely attributable to space; me, attributable to management and environment; ms, attributable to management and space; es, attributable to environment and space; mes, attributable to management, environment, and space; U, unexplained variation.

Results

Species Composition

The ordinations showed similar floristic relationships between treatments, regardless of whether planted occurrences were included. In Figure 2 we present the three-dimensional solution for wild occurrences only (planted individuals excluded), which was a substantially better fit to the data (stress 0.179) than the two-dimensional solution (stress 0.254). Remnant samples were clearly segregated from those of untreated pasture and restored vegetation, which were mixed with one another (Fig. 2). The linear vectors representing the target and observed trajectories for restoration were each significantly correlated with floristic composition ($r = 0.968$ and 0.903, respectively, both $p < 0.001$), but their directions were substantially divergent (Fig. 2). This suggests that changes in composition of the restored sites are not in the direction of increasing resemblance to remnant vegetation. When the restoration trajectories are viewed as lines connecting the average composition of each age class, they also show no evidence of convergence with the remnant reference sites (Fig. 2). The floristic segregation of remnants from revegetation and untreated pasture was maintained when the larger and more extensive sample of remnant vegetation was included in the ordination of cover-abundance data (Fig. 3).

There was a significant difference in species composition between all three management treatments when all species were included. However, differences between pasture and restored sites were slight and disappeared when planted occurrences were excluded from the analysis (Table 3). In contrast, differences between revegetated treatments and remnant vegetation were highly significant, irrespective of whether planted occurrences were excluded.

Twenty-five plant species contributed to 50% of the floristic dissimilarity between pasture and restored treatments in the "all species" analysis (Appendix). Only two of these (*Acacia parramattensis* and *Eucalyptus tereticornis*) were planted native species, but their exclusion rendered differences between pasture and restored treatments nonsignificant (Table 3). Eight introduced species (including five grasses, *Briza subaristata*, *Cynodon dactylon*, *Paspalum dilatatum*, *Phalaris minor*, and *Setaria gracilis*, and three forbs, *Plantago lanceolata*, *Senecio madagascariensis*, and *Sida rhombifolia*) and five native species (*Carex inversa*, *Geranium solanderi*, *Glycine clandestina*, *Microlaena stipoides*, and *Oxalis perennans*) shared high abundance across both pasture and restored treatments.

Thirty species contributed to 50% of floristic dissimilarity between restored and remnant vegetation, and 23 of these were also among the 29 species contributing to 50% of floristic dissimilarity between pasture and remnant vegetation (Appendix). Twenty of the 23 species that discriminated remnant vegetation from both pasture and restored vegetation were native trees, shrubs, forbs, and graminoids, and all but one (*Geranium solanderi*) were more abundant in remnant vegetation. Native species such as *Brunoniella australis*, *Corymbia maculata*, *Cheilanthes sieberi*, *Dillwynia sieberi*, two *Lomandra* spp., and *Panicum effusum* were abundant in remnant vegetation and either absent or extremely rare in pasture and restored vegetation. The three introduced species that discriminated remnant vegetation

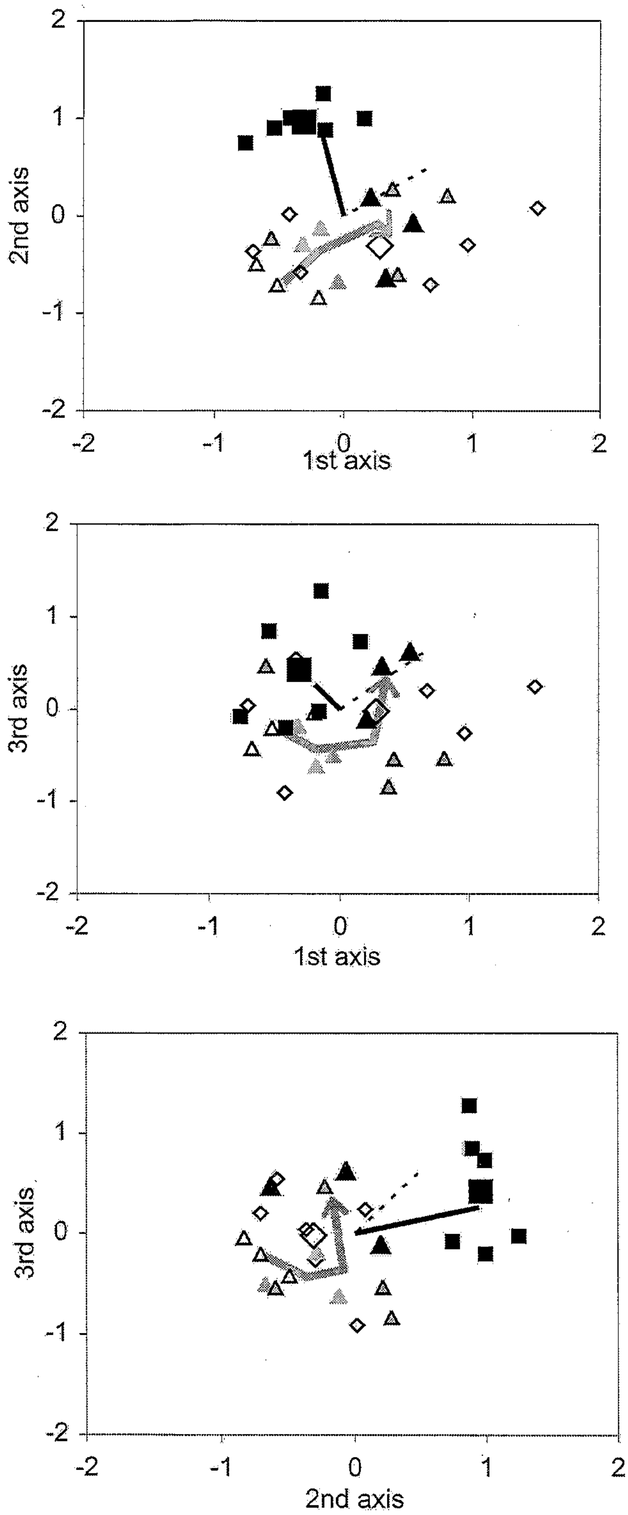


Figure 2. Multidimensional scaling ordination of floristic composition (frequency scores) excluding planted individuals for pasture (◇, control), revegetation of varying ages (Δ, 1 year; gray triangles with no outline, 3 year; gray triangles with black outline, 6 year; ▲, 9 year), and remnant vegetation (■, reference). The larger symbols represent the centroids (average position of pasture and remnant vegetation, respectively). Straight-line vectors represent the target trajectory from pasture to remnant (unbroken line) and the observed trajectory with increasing age of revegetation (broken line). Thick unbroken gray line joins the centroids of each revegetation age and represents the change in composition without assuming a straight-line trajectory.

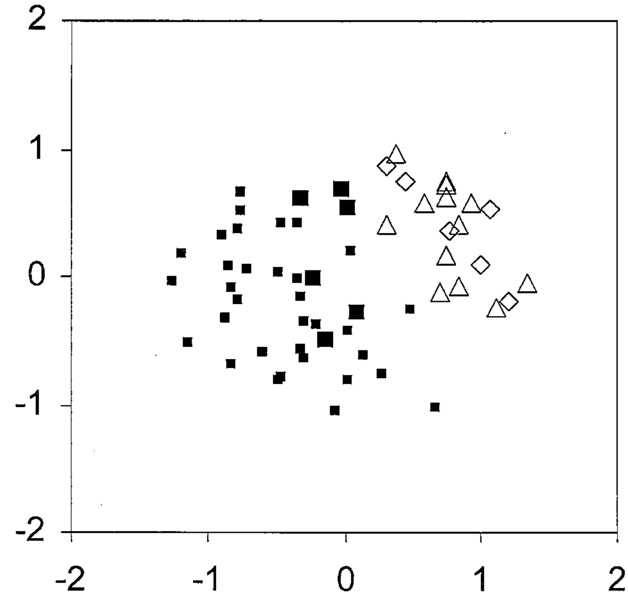


Figure 3. Multidimensional scaling ordination of floristic composition (cover-abundance estimates) excluding planted individuals for 25 samples of pasture (◇), revegetation (Δ), and remnant vegetation (■) collected in this study and an additional 33 samples of remnant vegetation (▪) from similar habitats within a 5-km radius of the study area. Stress = 0.213; only the first two of three axes are shown.

from both pasture and restored vegetation included two grasses, *Briza subaristata* and *Cynodon dactylon*, that were less abundant in remnant vegetation and the forb, *Senecio madagascariensis*, which was more abundant in remnant vegetation.

No significant differences in species composition were found between sites of different restoration age, and none of these differed from pasture (Table 4, $p > 0.05$). On the contrary, sites of all restoration ages differed significantly from remnant vegetation. This result was the same irrespective of whether planted occurrences were included in the analysis.

The two-way ANOSIM revealed no significant difference between the sites ($p > 0.1$) or management treatments, pasture cf. 6-year-old restoration ($p > 0.3$), irrespective of whether planted occurrences were included in the analysis. A one-way ANOSIM revealed no significant difference in species composition between the Hoxton Park and Prospect Reservoir remnant sites ($p > 0.1$).

Table 3. Analyses of similarity comparing floristic composition of vegetation subject to different management treatments.

Domain	*Management Treatment*	r *Statistic*	p *Value*
All vascular plants (including planted occurrences)	All management treatments	0.641	<0.001
	Pasture and restored	0.241	<0.05
	Pasture and remnant	0.83	<0.01
	Restored and remnant	0.953	<0.001
Wild occurrences only (excluding planted occurrences)	All management treatments	0.603	<0.001
	Pasture and restored	0.127	>0.1ns
	Pasture and remnant	0.830	<0.01
	Restored and remnant	0.972	<0.001

ns, not significant.

Sources of Variation

Canonical correspondence analyses showed that space (s), environment (e), and management (m) matrices explained 66–67% of the total variation in the floristic data, irrespective of whether planted individuals were included in the analysis (Fig. 4). We do not discuss planted individuals further because they had a negligible effect on the proportions of variation attributable to different sources.

Management alone accounted for only about 10% of total floristic variation. A further 16% of total variation was related to management and environment and/or space (components me, ms, and mse, Fig. 4). Management therefore contributed to 10–26% of total floristic variation. Within the management matrix treatment (pasture, restored, remnant) accounted for almost twice as much floristic variation as restoration age.

Both space and environment matrices accounted for larger portions of floristic variation than management. The space matrix alone accounted for the largest proportion (23%) of total variation (component s in Fig. 4), with an additional 18% of the total variation correlated with space and environment and/or management (components se + mse + ms, Fig. 4). The spatial variables therefore accounted for between 23 and 41% of total floristic variation. Within the space matrix site was the highest contributing indicator, whereas all indicators within the environment matrix contributed similar proportions of variation.

Table 4. Analyses of similarity to test for differences in species composition between individual age classes of restored vegetation, untreated pasture, and remnant vegetation.

Comparison	r *Statistic*	p *Value*
All classes	0.349	<0.001
Pasture cf. 1-year-old restored	−0.117	>0.6ns
Pasture cf. 3-year-old restored	−0.241	>0.9ns
Pasture cf. 6-year-old restored	−0.155	>0.8ns
Pasture cf. 9-year-old restored	−0.210	>0.8ns
1-year cf. 3-year-old restored	0.407	>0.1ns
1-year cf. 6-year-old restored	0.241	>0.2ns
1-year cf. 9-year-old restored	0.444	>0.1ns
3-year cf. 6-year-old restored	−0.056	>0.4ns
3-year cf. 9-year-old restored	0.241	>0.3ns
6-year cf. 9-year-old restored	−0.296	>0.8ns
1-year-old restored cf. remnant	0.963	<0.02
3-year-old restored cf. remnant	0.895	<0.02
6-year-old restored cf. remnant	0.952	<0.01
9-year-old restored cf. remnant	0.901	<0.02

Planted occurrences excluded. ns, not significant.

Community Structure

The pasture and restored treatments both had a significantly higher proportion of introduced species than remnant vegetation (Table 5), but pasture and restored treatments did not differ. These proportional differences were driven mainly by the larger number of native species in remnant vegetation, because differences in the number of introduced species between treatments were less pronounced (Table 5). There was no significant trend in the

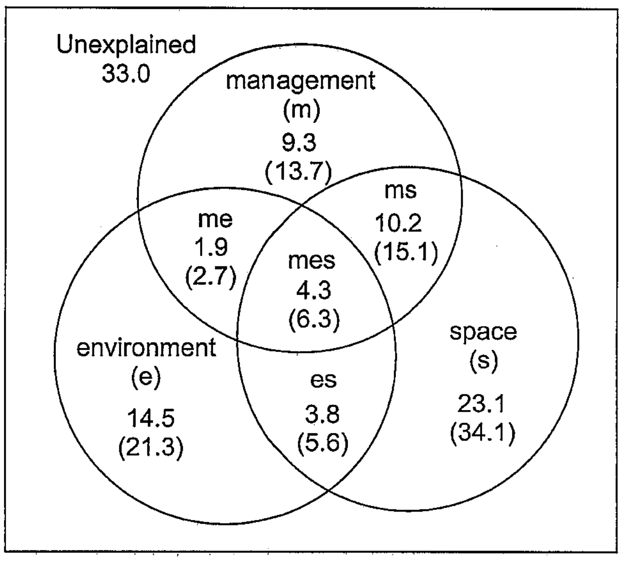


Figure 4. Venn diagram showing components of floristic variation attributable to management, environment, and space for wild occurrences of plant species (planted individuals excluded). Values are percentages of total variation with percentages of explained variation in parentheses.

Table 5. Structural characteristics of pasture, restored, and remnant vegetation (means with standard errors in parentheses).

	Pasture	*Restored*	*Remnant*	*F*	p
Proportion of introduced species	0.50 (0.06)[a]	0.50 (0.02)[a]	0.20 (0.02)[b]	28.82	<0.001
Number of native species	16.7 (3.5)[a]	16.2 (0.8)[a]	42.2 (3.2)[b]	36.3	<0.001
Number of exotic species	17.3 (2.1)[ab]	19.9 (1.2)[a]	13.7 (1.7)[b]	4.44	<0.05
Height of trees (m)	0.8 (0.2)[a]	5.2 (1.0)[b]	17.8 (1.5)[c]	49.62	<0.001
Cover of trees (%)	8 (2)[a]	26 (5)[b]	23 (2)[b]	11.56	<0.001
Cover of ground stratum (%)	94 (1)	88 (4)	88 (4)	1.2	>0.5ns
Cover of litter (%)	15.8 (2.0)	15.9 (2.4)	16.7 (1.7)	0.028	>0.9ns
Cover of bare soil (%)	4.2 (1.5)	9.6 (2.9)	5.8 (0.8)	1.10	>0.3ns

Means with the same superscripts are not significantly different at $p = 0.05$. The number of native species in the restored treatment excludes planted individuals (23.5 ± 0.7 when these are included). ns, not significant.

number of introduced species with age among the restored sites.

Tree foliage cover was significantly greater in the remnant and revegetated sites than in the pasture sites (Table 5), but there was no significant difference in the percentage canopy cover between remnant and revegetated sites. Regression indicated that both canopy height ($r^2 = 0.90$, $p < 0.001$) and crown cover ($r^2 = 0.51$, $p < 0.001$) increased significantly as restored vegetation aged. There was no significant difference in the percentage of ground cover between the management treatments (Table 5, $p >$ 0.5), but there was a weak trend of decreasing percentage ground cover with restoration age ($r^2 = 0.19, p < 0.05$).

Two-way analyses of variance revealed no differences in the number or proportion of introduced species between pasture and 6-year-old restored vegetation across two sites (Table 6). However, there was a significantly greater proportion of introduced species at Plough & Harrow than at Hoxton Park.

Canopy height differed significantly between these treatments but not between sites (Table 6). There was a significant interaction between site and treatment for ground cover, because the 6-year-old revegetated site had less gound cover than the pasture treatment at Plough and Harrow but not at Hoxton Park. Remnant vegetation did not vary in proportion or number of introduced species or any structural characteristics between the two sites at Hoxton Park and Prospect (all paired t-tests $p > 0.05$).

Table 6. Two-way analyses of variance or deviance of structural variables for two treatments (pasture and 6-year old restored vegetation) at two sites (Hoxton Park and Plough and Harrow property).

	Site	*Treatment*	*Interaction*
Proportion of introduced species	20.45*	1.95ns	0.54ns
Number of native species	8.17ns	0.02ns	1.25ns
Number of introduced species	1.39ns	0.82ns	2.47ns
Tree height	0.09ns	40.09**	0.82ns
Tree foliage cover	0.24ns	11.92*	0.68ns
Cover of ground stratum	13.02*		22.11**

$^{**}p < 0.01$, $^{*}p < 0.05$, ns not significant for each r value.

Discussion

Restoration Trajectory

Our floristic analyses show no clear evidence for a restoration trajectory from untreated pasture to remnant vegetation. The ordination suggested a weak trend in composition with age of restoration treatment, but this was in a different direction to the target trajectory from pasture to remnant vegetation. This conclusion holds irrespective of whether or not the restoration trajectory is assumed to be a linear trend through ordination space. The only floristic differences between the restored sites and untreated pasture detectable by ANOSIM were slight and due to the planted individuals. Similarly, the species richness data indicated that restored vegetation supported no more native species and no fewer exotic species than untreated pasture. On the contrary, restored vegetation had significantly more introduced species and less than half as many native species compared with remnant vegetation. Although several nonplanted native species were present in the restored areas, these appear to be species with opportunistic life histories that also persist in untreated pasture, sometimes at higher abundance than in remnant vegetation. Examples include *Asperula conferta* (Rubiaceae), *Carex inversa* (Cyperaceae), *Geranium solanderi* (Geraniaceae), *Oxalis perennans* (Oxalidaceae), and *Rumex brownii* (Polygonaceae). Such species seem to represent a low level of "habitat variegation" (McIntyre & Lavorel 1994) that is present in the landscape irrespective of restoration treatment. The restoration plantings and a decade of subsequent management therefore have not yet facilitated any significant unassisted recruitment of native plant species.

In contrast to the floristic analyses there is some evidence for structural development of restoration plantings, which have been shown to have very high rates of survival (92% over 3 years after planting; D. Williams, Greening Australia, 2002 unpublished data). There is strong evidence in our data that planted trees are growing and weak evidence that ground cover is thinning as the age of restoration plantings increases. This latter result may be the outcome of a competitive interaction between the growing tree stratum and the dense largely exotic ground cover of restored vegetation. Although we did not specifically examine ecosystem functions such as dispersal, recruitment, and competition, the floristic data suggest that dispersal, recruitment, or both processes are lacking in the restored sites relative to remnant vegetation (Clarke 2000).

Our results are tempered by limitations on sampling imposed by the original design of the restoration plantings. A more powerful analysis could be carried out if it were possible to implement a fully orthogonal replicated sampling design in which all the cells in Table 1 were filled. Sampling of the restored sites before treatment would further strengthen the comparisons (Buckney & Morrison 1995; Chapman & Underwood 2000).

The conclusion of no restoration trend seems to be robust for Hoxton Park, at which all three treatments and most restoration ages were sampled. The limited comparisons that were possible between sites indicate no differences between the same treatments or ages at different sites. Nevertheless, the limited replication across sites means that effects of management, including temporal trends, may have been partly confounded with spatial variation. Figure 4 (components ms and mse) indicates that about 15% of the total floristic variation may have been due to either management or spatial variation, compared with less than 10% attributable to management treatment alone (component m). These problems underscore the importance of designing a rigorous experimental monitoring program during the planning stage of restoration projects. Failure to do so seems to be a ubiquitous deficiency in restoration projects (Chapman & Underwood 2000), particularly considering that the project we studied was among the more thoroughly planned and best-resourced of those carried out in Australia.

Notwithstanding limitations imposed by sampling constraints, the floristic analyses support a steady-state model of vegetation dynamics in the restoration sites rather than directional succession (Connell & Slatyer 1977). The structural data may also be interpreted in support of this model if it is argued that the structural changes simply represent growth of planted individuals. Steady degraded states may persist because dysfunctional ecosystem processes maintain resources below threshold levels required to support many native species (Yates et al. 2000b). In this case space and light at ground level may be maintained below levels required for recruitment of native plants by the dense exotic ground cover.

Alternatively, the lack of compositional contrast between restored sites and untreated pasture and the lack of convergence of restored sites with remnant vegetation may be attributable to the comparatively short period of time elapsed since restoration was commenced. The significant trajectory observed in restoration treatments, together with the trend of reducing ground cover, offers some hope of floristic convergence in the longer term, even though development of the restoration treatments has not yet increased their floristic similarity to remnant vegetation. If the process of succession is very slow or if there is a significant lag time before recolonization and recruitment of native plant species is initiated, the available data cannot refute the possibility of succession over longer time frames in the order of several decades.

A number of ecological mechanisms may cause lags in successional change, including slow rates of seed dispersal, inhibition of seedling recruitment by dense exotic ground cover, residual allelopathic effects, and a dependence of seedling recruitment on fire. Although such traits are common in the Australian fire-prone flora (e.g., Keith et al. 2002), there are exceptions, particularly in grassy ecosystems such as the Cumberland Plain woodlands, which include some species with widely dispersed seeds and an ability to establish seedlings within small gaps in unburned vegetation (Clarke 2000; Lunt & Morgan 2002). If the dense exotic ground cover of restoration plantings inhibits recruitment of native seedlings, its treatment by herbicides during the establishment of plantings evidently did not mitigate its suppressive effect sufficiently or for long enough to permit substantial recruitment. Alterations to physical environmental characteristics of the sites during previous land uses, such as soil compaction, eutrophication, and landscape homogenization, could also be responsible for persistence in a steady state or very slow rates of succession (Yates et al. 2000a,b). Restoration sites at Hoxton Park, for example, had elevated soil nutrient levels relative to remnant vegetation (Perkins 1997). Resolution of these issues requires experimental manipulation and sampling over a longer restoration sequence than the 9 years available here.

Studies of other restored sclerophyllous plant communities have similarly failed to demonstrate an unambiguous trajectory in floristic composition from disturbed "control" samples to remnant "reference" samples. In a spatially and temporally replicated study Buckney and Morrison (1995) did detect a compositional trend over the first 15 years of postmining restoration of dry sclerophyll eucalypt forest on coastal sand dunes in southeastern Australia. However, as in our study they found that this trend was not in a direction of increasing similarity with vegetation that was on the mine site before mining and that the restored vegetation maintained its distinctiveness from adjacent unmined sites over 15 years. In a temperate New Zealand rainforest Reay and Norton (1999) detected no temporal trend in the total vascular plant composition of three restored sites (aged 12, 30, and 35 years) toward a mature forest site. However, they did detect trends of convergence in the tree flora and noted similarity in the composition of tree regeneration (i.e., individuals recruited after the original plantings) between the restored sites and an older naturally regenerating site. Nonetheless, the oldest regeneration site maintained its floristic distinctiveness from mature forest in both canopy trees and regenerating trees, as well as overall floristic composition. The comparative success of rainforest restoration compared with eucalypt forests and woodlands may be related to the dynamics of the exotic ground cover, which is thinned more effectively by a developing rainforest canopy than by sclerophyll trees and shrubs. These differences may be compounded by the greater shade tolerance of rainforest seedlings compared with those of sclerophyll species.

Campbell et al. (2002) found that restored wetlands in Pennsylvania, varying in age from 2 to approximately 20 years, supported fewer plant species than natural reference wetlands and maintained fundamentally different soil properties. The restored wetland flora was dominated by facultative annual species or clonal species, whereas a number of wetland specialists were less abundant than in the reference wetlands. Zedler and Callaway (1999) also concluded that several ecological attributes of a restored Californian wetland failed to converge with those of a natural wetland over 11 years of observation. They were consequently critical of models that predict a smooth trajectory of restored sites, which rapidly converges with natural reference sites.

Implications for Restoration and Mitigation Policies

Although studies of restoration success for native vegetation are still few in number, a common conclusion emerges from this study and others in different ecosystems. The development of species composition in restored sites toward a state that resembles appropriate reference sites is, at best, extremely slow and may not eventuate at all. If such a succession occurs the time scales required for restored sites to match the target state range from several decades and may extend to the order of centuries. The conclusion is even more sobering, considering that our study examined a well-resourced well-documented restoration project that was implemented by a very dedicated workforce. Still more alarming is the likelihood that the outcomes of administrative audits based on numbers of dollars spent, hectares planted, or volunteers engaged may be misinterpreted as signals of restoration success in the absence of a satisfactory ecological audit. Clearly, the conspicuous absence or inadequacy of ecological audits in restoration projects needs an urgent remedy.

The limited success of restoration projects should not be interpreted as a reason for their abandonment. On the contrary, studies such as ours underscore the necessity to develop improved methods and resourcing for ecosystem restoration and point to the need for a better balance of priorities between restoration and protection of remnant native ecosystems.

Our conclusions have important implications for policies governing the regulation of development and the management of ecosystems and their biological diversity. This is especially so for policies involving "mitigation," "offsets," "credits," or "no net loss," whereby approvals for developments that destroy or degrade a natural asset are predicated on undertakings to carry out compensatory actions elsewhere, such as restoration of degraded ecosystems or reconstruction of habitat. One such policy has governed regulation of wetland development for some years in the United States (Environmental Protection Agency and Department of the Army 1990), whereas similar policies concerning the regulation of clearing of native vegetation are currently under consideration by state and Commonwealth governments in Australia.

Limitations on both the success and the rates of ecosystem restoration observed in this study support recommendations for regulatory approaches that seek to prevent damages to ecosystems rather than those that permit losses and hope for compensation (Zedler & Callaway 1999). Zedler and Callaway (1999) suggested that mitigation policies should include recognition that compensation sites may never fully replace natural sites and that the time required for restoration may exceed traditional expectations and planning horizons. However, weighted compensation ratios (where the restored area required exceeds the loss) can only partly deal with these problems, which must be dealt with in a broader context that includes modifying design of developments to reduce or avoid impacts and refusal of development approvals. Policies that seek to balance or overcompensate losses of biodiversity with gains are fundamentally flawed if there is no feasible restoration technology to achieve replacement.

Although restoration will always have an essential role in biodiversity conservation, our results indicate that a much higher premium needs to be placed on native ecosystems that recognizes both the difficulty and cost of their replacement or, arguably, recognizes their irreplaceability. Regulatory policies also need to incorporate long-term commitments to monitoring for ecological audits that include clear restoration goals and a design that enables experimental evaluation in a rigorously controlled and replicated manner (Chapman & Underwood 2000).

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Appendix Abundance of vascular plant species and their percentage contribution to dissimilarities between three management treatments.

Family and Species	Mean Frequency Score (se)			% Contribution to Dissimilarity (Top 50% Cumulative Contribution)		
	Pasture	Revegetated	Remnant	Pasture cf. Revegetated	Pasture cf. Remnant	Revegetated cf. Remnant
Ferns						
Sinopteridaceae						
Cheilanthes sieberi	0 (0)	0 (0)	2.50 (2.17)		1.24	1.22
Dicotyledons						
Acanthaceae						
Brunoniella australis	0.33 (0.33)	0.23 (0.34)	6 (0)		3.16	3.13
Amaranthaceae						
Gomphrena celosioides[a]	0.50 (0.50)	0 (0)	0 (0)			
Apiaceae						
Centella asiatica	0.67 (0.49)	0.23 (0.24)	1.83 (2.48)			
Cyclospermum leptophyllum[a]	0 (0)	0.31 (0.26)	0.17 (0.41)			
Foeniculum vulgare[a]	0 (0)	0.15 (0.23)	0 (0)			
Hydrocotyle peduncularis	0 (0)	0 (0)	0.50 (0.84)			
Trachymene incisa	0.17 (0.17)	0.69 (0.51)	0 (0)			
Asclepiadaceae						
Araujia sericeum[a]	0 (0)	0.31 (0.2)	0.33 (0.52)			
Gomphocarpus fruticosus[a]	0 (0)	0.08 (0.11)	0.17 (0.41)			
Asteraceae						
Aster subulatus[a]	0.17 (0.17)	0 (0)	0 (0)			
Bidens pilosa[a]	1.00 (0.82)	0.31 (0.35)	1.00 (1.26)			
Cirsium vulgare[a]	1.50 (0.76)	1.92 (0.70)	1.67 (1.97)	1.55		
Conyza albida[a]	0 (0)	1.00 (0.67)	0.83 (0.98)			
Conyza canadensis[a]	0 (0)	0 (0)	0.17 (0.41)			
Euchiton gymnocephalus	0 (0)	0.15 (0.15)	0.17 (0.41)			
Hypochaeris radicata[a]	1.00 (1.00)	0.69 (0.48)	1.33 (1.51)			
Onopordum acanthium ssp. *acanthium*[a]	0.50 (0.34)	0.46 (0.39)	1.17 (1.17)			
Senecio madagascariensis[a]	2.17 (1.05)	3.00 (0.93)	4.5 (2.26)	2.32	1.80	1.44
Senecio pterophorus[a]	0 (0)	0.08 (0.11)	0 (0)			
Senecio quadridentatus	0 (0)	0 (0)	0.33 (0.82)			
Sonchus asper ssp. *glaucescens*[a]	0 (0)	0 (0)	0.50 (0.84)			
Sonchus oleraceus[a]	0.67 (0.33)	1.23 (0.75)	0.83 (0.41)			
Sonchus spp.	0 (0)	0.62 (0.61)	0 (0)			
Taraxacum officinale[a]	0.67 (0.67)	0.23 (0.24)	0.33 (0.52)			
Vernonia cinerea	0 (0)	0 (0)	1.50 (1.87)			
Brassicaceae						
Brassica juncea[a]	0 (0)	0.08 (0.11)	0 (0)			
Brassica rapa ssp. *sylvestris*[a]	0 (0)	0.15 (0.23)	0 (0)			
Cactaceae						
Opuntia stricta var. stricta[a]	0 (0)	0 (0)	0.17 (0.41)			
Campanulaceae						
Wahlenbergia gracilis	0.17 (0.17)	0 (0)	0.50 (0.84)			
Caryophyllaceae						
Cerastium glomeratum[a]	0.33 (0.21)	0.23 (0.24)	0 (0)			
Chenopodiaceae						
Einadia hastata	0.17 (0.17)	0.08 (0.11)	0 (0)			
Convolvulaceae						
Convolvulus erubescens	0.17 (0.17)	0 (0)	0.17 (0.41)			
Dichondra repens	1.00 (0.37)	2.00 (0.9)	5.67 (0.52)	1.64	2.55	1.99
Epacridaceae						
Leucopogon juniperinus	0.17 (0.17)	0 (0)	0 (0)			
Euphorbiaceae						
Phyllanthus virgatus	0.17 (0.17)	0.15 (0.15)	1.17 (1.6)			
Fabaceae						
Acacia falcata[b]	0 (0)	1 (0.6)	0 (0)			
Acacia implexa[b]	0.17 (0.17)	0.77 (0.56)	0.5 (0.55)			
Acacia parramattensis[c]	0 (0)	2.85 (0.52)	0.17 (0.41)	2.49		1.45

(*Continued*)

Appendix (*Continued*)

Family and Species	*Mean Frequency Score (se)* Pasture	Revegetated	Remnant	*% Contribution to Dissimilarity (Top 50% Cumulative Contribution)* Pasture cf. Revegetated	Pasture cf. Remnant	Revegetated cf. Remnant
Fabaceae (*Continued*)						
Bossiaea prostrata	0 (0)	0 (0)	0.50 (0.84)			
Daviesia genistifolia	0 (0)	0.15 (0.23)	0 (0)			
Daviesia ulicifolia[b]	0 (0)	0.23 (0.24)	0.83 (1.17)			
Desmodium rhytidophyllum	0 (0)	0 (0)	0.17 (0.41)			
Desmodium varians	0 (0)	0.54 (0.49)	2.17 (2.40)			
Dillwynia sieberi[c]	0 (0)	0.23 (0.18)	3.17 (0.75)		1.77	1.60
Fabaceae sp.	0 (0)	0.08 (0.11)	0 (0)			
Glycine clandestina	1.17 (0.65)	2.00 (0.53)	3.83 (2.14)	1.49	1.64	1.27
Glycine microphylla	0.67 (0.49)	0.62 (0.49)	1.33 (1.51)			
Glycine tabacina	3.33 (1.15)	4.54 (0.70)	5.33 (1.03)	2.31	1.47	
Hardenbergia violacea[b]	0 (0)	0.23 (0.24)	2.00 (2.53)		1.28	1.21
Indigofera australis[c]	0.17 (0.17)	0.46 (0.32)	0.33 (0.82)			
Lotus angustissimus[a]	1.33 (0.95)	2.31 (0.75)	0.17 (0.41)	1.96		1.18
Pultenaea microphylla[b]	0 (0)	0.15 (0.15)	1.50 (1.64)			
Trifolium dubium[a]	0.17 (0.17)	0.31 (0.20)	0 (0)			
Trifolium medicago[a]	0 (0)	0.46 (0.46)	0 (0)			
Trifolium repens[a]	0.50 (0.50)	0.31 (0.20)	0 (0)			
Trifolium striatum[a]	0 (0)	0.08 (0.11)	0 (0)			
Vicia sativa ssp. *sativa*[a]	0 (0)	0.08 (0.11)	0 (0)			
Gentianaceae						
Centaurium tenuiflorum[a]	0 (0)	0.08 (0.11)	0 (0)			
Geraniaceae						
Geranium homeanum	0.17 (0.17)	0.23 (0.24)	0 (0)			
Geranium solanderi	2.83 (1.01)	3.38 (0.95)	0.67 (0.82)	2.28	1.41	1.60
Pelargonium inodorum	0.50 (0.34)	0.85 (0.66)	0 (0)			
Goodeniaceae						
Goodenia hederacea ssp. *hederacea*	0 (0)	0 (0)	1.33 (1.51)			
Hypericaceae						
Hypericum gramineum	1.67 (0.95)	1.46 (0.57)	1.33 (1.51)	1.61		
Lamiaceae						
Ajuga australis	0 (0)	0 (0)	0.83 (1.33)			
Mentha satureioides	0 (0)	0 (0)	0.50 (0.84)			
Linaceae						
Linum marginale	0 (0)	0 (0)	0.33 (0.82)			
Lobeliacaeae						
Pratia purpurascens	0.33 (0.33)	0 (0)	2.17 (2.14)			
Malvaceae						
Sida rhombifolia[a]	2.83 (0.48)	4.62 (0.57)	2.17 (1.94)	1.80		1.56
Myoporaceae						
Eremophila debilis	0 (0)	0 (0)	1.17 (1.6)			
Myrtaceae						
Angophora floribunda[b]	0 (0)	0.38 (0.27)	0 (0)			
Angophora subvelutina[b]	0 (0)	0.54 (0.32)	0 (0)			
Corymbia maculata[c]	0 (0)	0.31 (0.26)	3 (1.1)		1.69	1.49
Eucalyptus amplifolia[b]	0.17 (0.17)	0.46 (0.36)	0.17 (0.41)			
Eucalyptus baueriana[b]	0 (0)	0.15 (0.23)	0 (0)			
Eucalyptus crebra[b]	0 (0)	0.23 (0.24)	1.17 (1.60)			
Eucalyptus eugenioides[b]	0 (0)	0.08 (0.11)	0.50 (0.84)			
Eucalyptus globoidea[b]	0 (0)	0.08 (0.11)	0 (0)			
Eucalyptus moluccana[c]	0 (0)	1.46 (0.43)	2.33 (1.21)		1.32	
Eucalyptus tereticornis[b]	0.17 (0.17)	2.08 (0.61)	1.50 (1.52)	1.66		
Melaleuca decora[c]	0 (0)	0.15 (0.15)	0 (0)			
Melaleuca linariifolia[b]	0 (0)	0.31 (0.31)	0 (0)			
Melaleuca styphelioides[c]	0 (0)	0.31 (0.26)	0 (0)			
Oleaceae						
Olea europaea ssp. *africana*[a]	0 (0)	0.31 (0.35)	0.17 (0.41)			

(*Continued*)

Appendix *(Continued)*

Family and Species	*Mean Frequency Score (se)* Pasture	Revegetated	Remnant	*% Contribution to Dissimilarity (Top 50% Cumulative Contribution)* Pasture cf. Revegetated	Pasture cf. Remnant	Revegetated cf. Remnant
Onagraceae						
Epilobium billardierianum ssp. *cinereum*	0 (0)	0.15 (0.23)	0 (0)			
Oxalidaceae						
Oxalis perennans	2.5 (0.72)	3.92 (0.75)	1.67 (1.63)	2.00		1.53
Pittosporaceae						
Billardiera scandens	0 (0)	0.08 (0.11)	0 (0)			
Bursaria spinosa[c]	0 (0)	0.85 (0.62)	2.33 (1.97)		1.16	
Plantaginaceae						
Plantago debilis	0 (0)	0.08 (0.11)	0.17 (0.41)			
Plantago gaudichaudii	0 (0)	0 (0)	0.33 (0.82)			
Plantago lanceolata[a]	2.33 (0.67)	3.31 (0.86)	1.50 (1.52)	1.91		1.35
Polygonaceae						
Rumex brownii	1.17 (0.48)	1.08 (0.46)	0.67 (1.03)			
Rumex crispus[a]	0.50 (0.50)	0.54 (0.68)	0 (0)			
Primulaceae						
Anagallis arvensis[a]	0.83 (0.54)	2.08 (0.74)	0.17 (0.41)	1.68		
Proteaceae						
Grevillea robusta[a]	0 (0)	0 (0)	0.17 (0.41)			
Ranunculaceae						
Ranunculus lappaceus	0.83 (0.65)	0.38 (0.31)	0.67 (0.82)			
Rosaceae						
Rubus leightoni[a]	0 (0)	0.15 (0.15)	0 (0)			
Rubiaceae						
Asperula conferta	2.00 (1.13)	1.77 (0.99)	0.50 (0.84)	2.11		
Opercularia aspera	0 (0)	0 (0)	0.50 (1.22)			
Opercularia diphylla	0 (0)	0 (0)	0.33 (0.52)			
Opercularia varia	0 (0)	0 (0)	1.00 (2.45)			
Richardia stellaris[a]	0 (0)	0 (0)	0.33 (0.52)			
Sherardia arvensis[a]	0.17 (0.17)	0.38 (0.36)	0.17 (0.41)			
Sapindaceae						
Dodonaea viscosa ssp. *cuneata*[a]	0 (0)	0.77 (0.41)	0.83 (1.33)			
Scrophulariaceae						
Veronica brownii	0.17 (0.17)	0 (0)	0.83 (2.04)			
Veronica plebeia	0 (0)	0 (0)	0.17 (0.41)			
Solanaceae						
Cestrum parqui[a]	0.17 (0.17)	0.23 (0.24)	0 (0)			
Lycium ferocissimum[a]	0 (0)	0.08 (0.11)	0 (0)			
Nicotiana debneyi[a]	0 (0)	0.08 (0.11)	0 (0)			
Solanum nigrum[a]	0.67 (0.49)	0.38 (0.27)	0.50 (0.55)			
Solanum physalifolium[a]	0 (0)	0.08 (0.11)	0 (0)			
Solanum pungetium	0 (0)	0.08 (0.11)	0.83 (0.75)			
Solanum spp.	0 (0)	0 (0)	0.17 (0.41)			
Verbenaceae						
Verbena incompta[a]	1.67 (0.56)	2.23 (0.71)	0 (0)	1.45		1.22
Verbena littoralis[a]	2.67 (0.99)	0.69 (0.51)	0 (0)	2.30	1.58	
Verbena rigida[a]	2.00 (0.86)	1.85 (0.62)	0.33 (0.52)	1.69		
Monocotyledons						
Anthericaceae						
Laxmannia gracilis	0.33 (0.21)	0.23 (0.18)	0 (0)			
Arthropodium milleflorum	0 (0)	0 (0)	1.67 (1.63)			
Caesia parviflora var. *vittata*	0 (0)	0 (0)	0.33 (0.82)			
Asparagaceae						
Myrsiphyllum asparagoides[a]	0 (0)	0.23 (0.18)	0 (0)			
Commelaceae						
Commelina cyanea	0 (0)	0.08 (0.11)	0 (0)			
Cyperaceae						
Carex breviculmis	0 (0)	0 (0)	0.17 (0.41)			

(Continued)

Appendix (*Continued*)

Family and Species	Mean Frequency Score (se) Pasture	Mean Frequency Score (se) Revegetated	Mean Frequency Score (se) Remnant	% Contribution to Dissimilarity (Top 50% Cumulative Contribution) Pasture cf. Revegetated	Pasture cf. Remnant	Revegetated cf. Remnant
Cyperaceae (*Continued*)						
Carex inversa	3.17 (0.65)	3.23 (0.77)	2.00 (2.19)	1.67	1.27	1.35
Cyperus brevifolius	0 (0)	0.31 (0.45)	0 (0)			
Cyperus gracilis	0 (0)	0.62 (0.57)	1.33 (2.42)			
Cyperus sesquiflorus	0 (0)	0.08 (0.11)	0.17 (0.41)			
Fimbristylis dichotoma	0 (0)	0.15 (0.23)	0.67 (1.21)			
Hypoxidaceae						
Hypoxis pratensis	0 (0)	0 (0)	0.33 (0.52)			
Iridaceae						
Freesia sp. (hybrid)[a]	0.17 (0.17)	0 (0)	0 (0)			
Gladolius gueinzii[a]	0.17 (0.17)	0.08 (0.11)	0 (0)			
Juncaceae						
Juncus usitatus	1.67 (0.61)	0.38 (0.31)	0 (0)			
Lomandraceae						
Lomandra confertifolia ssp. *rubiginosa*	0 (0)	0 (0)	2.83 (1.94)		1.57	1.53
Lomandra longifolia[b]	0 (0)	0.77 (0.56)	1.5 (1.38)			
Lomandra multiflora	0 (0)	0 (0)	2.33 (2.07)		1.42	1.37
Phormiacae						
Dianella longifolia var. *longifolia*[b]	0.5 (0.34)	0.08 (0.11)	3.17 (2.14)		1.63	1.70
Poaceae						
Aristida ramosa	0.83 (0.54)	0.54 (0.59)	3.83 (2.56)		1.80	1.89
Aristida vagans	0.5 (0.34)	0.85 (0.6)	2.67 (2.73)		1.26	1.28
Austrodanthenia racemosa	0.33 (0.21)	0.15 (0.15)	0.17 (0.41)			
Axonopus affinis[a]	0.17 (0.17)	0.54 (0.59)	1.33 (1.97)			
Bothriochloa decipiens	0 (0)	0.54 (0.39)	1.50 (1.76)			
Briza maxima[a]	1.17 (0.6)	0.92 (0.61)	0 (0)			
Briza minor[a]	0 (0)	0.23 (0.24)	0.17 (0.41)			
Briza subaristata[a]	3.33 (0.8)	3.46 (0.97)	0.17 (0.41)	2.14	1.73	1.81
Bromus catharticus[a]	1.17 (0.75)	0.31 (0.26)	0 (0)			
Chloris gayana[a]	0.50 (0.22)	0.31 (0.26)	0.17 (0.41)			
Chloris truncata	0.17 (0.17)	0.54 (0.57)	1.67 (2.34)			
Cymbopogon refractus	0.33 (0.33)	0.38 (0.36)	3.00 (2.19)		1.43	1.41
Cynodon dactylon[a]	4.67 (0.71)	4.69 (0.73)	0 (0)	1.54	2.55	2.55
Dactyloctenium radulans	0 (0)	0 (0)	0.17 (0.41)			
Dichanthium sericeum	1.33 (0.61)	1.08 (0.72)	0.33 (0.52)			
Dichelachne micrantha	1.00 (0.52)	1.00 (0.53)	1.83 (1.94)			
Digitaria ramularis	0 (0)	0 (0)	0.33 (0.52)			
Echinopogon ovatus	0.33 (0.21)	0.31 (0.35)	2.83 (2.23)		1.41	1.38
Elymus scaber	0.83 (0.48)	0.77 (0.56)	0 (0)			
Enteropogon acicularis	0.17 (0.17)	0.08 (0.11)	0 (0)			
Entolasia marginata	0 (0)	0.08 (0.11)	0.50 (0.84)			
Eragrostis elongata	2.17 (0.95)	2.31 (0.87)	2.83 (3.13)	1.99	1.53	1.53
Eragrostis leptostachya	0 (0)	0.23 (0.34)	0.5 (0.84)			
Eriochloa pseudoacrotricha	0.17 (0.17)	0 (0)	0 (0)			
Lolium perenne[a]	0 (0)	0.08 (0.11)	0 (0)			
Microlaena stipoides	3.67 (0.84)	4.85 (0.57)	5.67 (0.82)	1.86	1.26	
Nassella neesiana[a]	0 (0)	0.08 (0.11)	0 (0)			
Panicum effusum	0.17 (0.17)	0.23 (0.34)	4.50 (0.84)		2.40	2.29
Paspalidium gracile	0 (0)	0 (0)	1.50 (2.35)			
Paspalum dilatatum[a]	5.17 (0.48)	5.69 (0.26)	2.67 (2.8)		1.81	1.89
Pennisetum clandestinum[a]	3.00 (1.34)	0.23 (0.24)	0 (0)	2.79	1.78	
Phalaris minor[a]	2.17 (1.22)	3.38 (1.01)	0.17 (0.41)	2.61		1.75
Setaria gracilis[a]	4.17 (0.79)	5.38 (0.49)	4.17 (1.94)	1.70		
Sporobolus africanus[a]	1.17 (0.98)	0.62 (0.49)	0.67 (1.21)			
Sporobolus creber	0.17 (0.17)	0.15 (0.15)	0.50 (0.84)			
Sporobolus elongatus	0.33 (0.33)	0.92 (0.61)	3.17 (2.14)		1.53	1.42
Themeda australis	1.33 (0.95)	0.85 (0.47)	4.33 (1.97)		2.09	2.06

Data include planted occurrences.
[a] Introduced species.
[b] Species represented in the restored treatment by planted individuals only.
[c] Species represented in the restored treatment by both planted and nonplanted individuals.