LANDSCAPE LOGIC

LINKING LAND AND WATER MANAGEMENT TO RESOURCE CONDITION TARGETS

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Detecting short-term change in the structure and composition of dry grassy forest

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LANDSCAPE LOGIC is a research hub under the Commonwealth Environmental Research Facilities scheme, managed by the Department of Environment, Water Heritage and the Arts. It is a partnership between:

- **six regional organisations** – the North Central, North East & Goulburn–Broken Catchment Management Authorities in Victoria and the North, South and Cradle Coast Natural Resource Management organisations in Tasmania;
- **five research institutions** – University of Tasmania, Australian National University, RMIT University, Charles Sturt University and CSIRO; and
- **state land management agencies in Tasmania and Victoria** – the Tasmanian Department of Primary Industries & Water, Forestry Tasmania and the Victorian Department of Sustainability & Environment.

The purpose of Landscape Logic is to work in partnership with regional natural resource managers to develop decision-making approaches that improve the effectiveness of environmental management.

Landscape Logic aims to:

1. Develop better ways to organise existing knowledge and assumptions about links between land and water management and environmental outcomes.
2. Improve our understanding of the links between land management and environmental outcomes through historical studies of private and public investment into water quality and native vegetation condition.

Detecting short-term change in the structure and composition of dry grassy forest

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Summary

Natural resource managers worldwide are attempting to protect and restore vegetation to conserve biodiversity and ensure supply of ecosystem services. They typically measure progress toward this objective using generic and uncertain assumptions about the way vegetation attributes change under different management regimes over time.

The ability to develop more accurate predictive models of vegetation change as a guide to management is commonly limited by the availability of sensitive baseline data. But despite the widespread recognition of the value of monitoring data, there are very few examples where data of sufficient sensitivity is collected.

In this study, we repeated measurements five years after detailed data on habitat quality had been collected from maturing grassy dry Eucalypt forest vegetation in the peri-urban fringe north east of Melbourne, Australia. The assessment included mature tree density, tree canopy cover, tree species recruitment, and abundance and diversity of understorey species and life forms.

The aims were:

1. To determine whether change in vegetation composition and structure could be detected in a maturing dry forest within a five-year period.
2. To determine whether the sampling design and intensity was adequate to detect changes in cover of different life form classes within sites, particularly those with low or variable cover.

In the absence of fire, consistent changes in both structure and composition were detected over the five-year period with increases detected in recruitment of *Eucalyptus spp.*, density of dead stems and cover of small logs, litter and soil crust. Decline was commonly observed in the cover of understorey components and overall species richness, which are attributes most likely affected by prolonged dry conditions.

At sites subject to fire, changes were more marked with declines detected in species richness as well as cover of bryophytes, soil crust, shrubs, medium tufted graminoids and herbs. In contrast, bare ground cover increased over the sampling period.

The method was found to be sufficiently sensitive to detect relatively subtle changes and may assist our understanding of native vegetation response to management.

Changes in the structure and composition of this forest type detected in this study can be plausibly linked to climate and fire. However, it is not possible to determine whether these changes are directional or indicate normal cyclical fluctuations. Until the nature and magnitude of the variability inherent in vegetation change is better understood, managers will not be in a position to identify change in these attributes early enough to successfully intervene.

Contents

Introduction

Globally, natural resource and land management agencies are trying to protect and restore indigenous vegetation for biodiversity and the supply of ecosystem services (Saunders *et al.* 1991; Millenium Ecosystem Assessment 2005; Turner *et al.* 2007; Lovett *et al.* 2008). At regional scales, assessing and reporting progress toward this objective is typically based on generic assumptions about the way that vegetation attributes change under different management regimes over time (Manning *et al.* 2006; DSE 2008b). These assumptions are often highly uncertain (Dorrough *et al.* 2008; Duncan and Wintle 2008; Vesk *et al.* 2008), partly because of the inherent complexity of ecological processes and patterns (Holling 1996). Improved knowledge about the trajectory of native vegetation change is required if managers are to intervene successfully to achieve gains in native vegetation extent and quality (Gibbons *et al.* 2006; DSE 2008b; Thackway and Lesslie 2008).

Our ability to develop quantitative models of vegetation change is often limited by the availability of sensitive baseline data (Taverna *et al.* 2005). Despite the widespread appreciation of the value of monitoring data to support adaptive learning about management of native vegetation, there are precious few examples where appropriate data are collected. Within the mining industry, the introduction of environmental bonds to encourage better practice in the ecological restoration of mine sites led to methods and data to evaluate success (Koch and Hobbs 2007). However, restoration of mine sites is comparatively uncomplicated to assess. There are fewer sites, they start from extremely degraded states, and changes in ecological process occur relatively quickly. By contrast, public investment in native vegetation management has less money available, many more sites and land managers, less gross change expected, and no equivalent imperative to demonstrate management performance. In Australia, recent audits of major national native vegetation investment programs concluded that there was little evidence to support their effectiveness in improving habitat quality (ANAO 2008). Meanwhile there is increasing pressure from government to demonstrate environmental benefits from the investment of public funds in native vegetation management.

Currently there are several methods available for assessing vegetation quality for conservation planning and investment purposes (DSE 2004; Gibbons *et al.* 2005) However, there have been very few demonstrations of sampling methods capable of detecting change in the native vegetation attributes widely used in habitat quality assessments. Consequently we are unable to demonstrate change as a result of management interventions.

In this study, we revisited maturing grassy dry forest vegetation sites from which detailed habitat quality data was collected in 2002/2003 (Tolsma and Newell 2003). These assessments included mature tree density, tree canopy cover, tree species recruitment, abundance and diversity of understorey species and life forms. These are considered important indicators of vegetation condition because they represent measures of community composition, structure and function (Gibbons and Freudenberger 2006b; McElhinny *et al.* 2006), and are presumed to play an integral role in influencing biodiversity and habitat diversity (Parkes *et al.* 2003b).

Understanding change in maturing vegetation is important because relatively mature and unmodified vegetation is often used as a benchmark for the condition or habitat value of vegetation in conservation planning contexts (Parkes *et al.* 2003a; Gibbons and Freudenberger 2006b; Gibbons *et al.* 2008). Thus, an understanding of the inherent variability and change within 'reference' sites is required, as these desirable attributes may be shifting targets (Taverna *et al.* 2005; Gibbons and Freudenberger 2006a).

The rationale for this research was that if change could be detected in these maturing vegetation systems over a relatively short period, the method may enable us to learn about vegetation change in other more dynamic contexts.

The aims were:

1. To determine whether change in vegetation composition and structure could be detected in a maturing dry forest within a five-year period.
2. To determine whether the sampling design and intensity was adequate to detect changes in cover of different life form classes within sites, particularly those with low or variable cover.

Methods

Site Selection

Between March 2008 and April 2008, 18 of the 41 sites sampled by Tolsma and Newell (2003) were revisited. These sites were all within the Ecological Vegetation Class (EVC) 'Grassy Dry Forest' (EVC 22, DSE 2008a) and were situated north-east of Melbourne on public land; in the bio-region Highlands–Southern Fall, Victoria (Figure 1, Table 1).

This EVC group occurs throughout Victoria over a range of substrates and altitudes (DSE 2008a). Rainfall across the study area is variable, ranging from 700–1000mm per year (BoM 2008). The mostly uneven aged stand structure at these sites had an average stem density of 313 (± 81) mature stems per 0.4 ha quadrat. The overstorey was predominately *Eucalyptus macrorhyncha F.Muell. ex Benth, E. goniocalyx F.Muell. ex Miq.,* and *E. polyanthemos Schauer,* interspersed with other *Eucalyptus* species and *Exocarpus cupressiformis Labill.* Where present, a secondary tree or tall-shrub layer was dominated by *Acacia* species. The understorey comprised a diversity of life forms, generally consisting of an open shrub layer overlying a grassy understorey with a sparse cover of low-lying shrubs and herbs, and a high litter component.

The 18 sites selected for re-survey were those with the lowest and highest values for cover of exotic species, bare ground, tree canopy, and stand age (Table 1). Mapping of fire events (DSE 2008b) showed that five of the sites had been burnt in the intervening period between 2002 and 2008. Of the other 13 sites, fire mapping indicated that one had been burnt in the five years prior to 2002, whilst the remainder had not been burnt since 1990. Although our study design was not concerned specifically with the effects of fire, we have used the data from the five burnt sites as an out-group for comparative purposes.

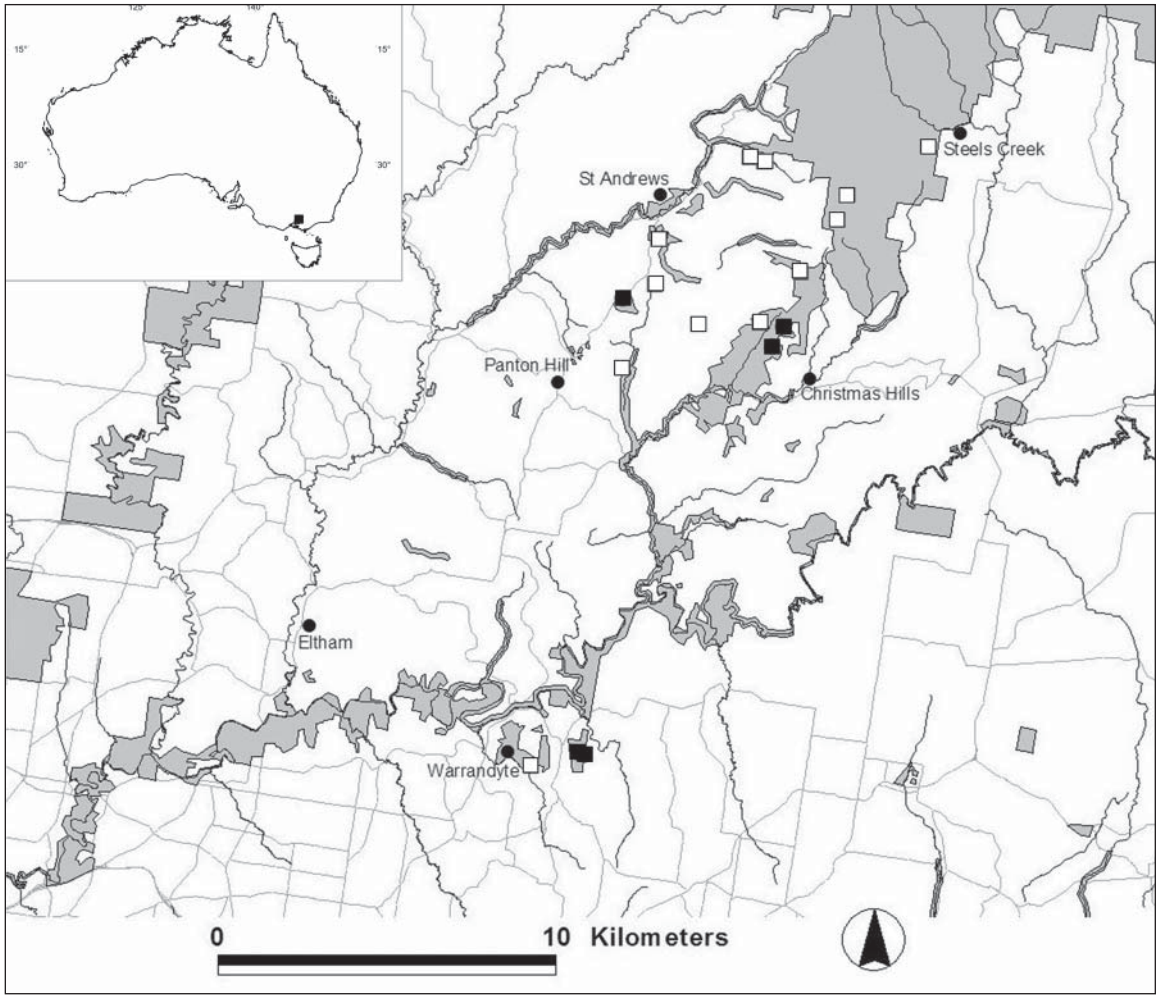


Figure 1. Location of study sites to the north-east of Melbourne, Victoria, Australia (inset). Filled squares indicate sites that were burned between 2002 and 2008, open squares are unburned sites. Also shown are public land blocks, roads and creeks..

Sampling methods

The sampling method directly followed that used by Tolsma & Newell (2003). The approximate position of each sampling plot was relocated using a GPS, and then a more precise location was established by referring to transect photos taken in 2002. At each site a 100m transect was established using the same starting point and direction used in 2002/03. Flags were placed at regular intervals either side of this transect at 20m perpendicular to the transect to delineate a quadrat measuring 100m x 40m (Figure 2,A). Within this quadrat, the following components were measured.

- Tree Canopy Cover. Ten digital images were taken at 10m intervals along the 100m long central transect, using a tripod-mounted Canon S40 digital camera (4.0 megapixels, 35mm focal length) directed vertically upwards. The pixels in each image were later allocated to either canopy or sky on the basis of colour, using the proprietary software WINCAM (Regent Instruments 2002), to derive a value for average projective canopy cover.
- Large Trees. The number of trees >50cm DBH within the quadrat were recorded.
- Logs. The lengths and diameter of all logs (>10cm diameter) within the quadrat were measured using Vernier callipers and a tape measure. Their length (or portion thereof) were placed into one of 10 size classes (10–20, 21–30, 31–40, 41–50, 51–60, 61–70, 71–80, 81–90, 91–100, >100cm).
- Understorey structure. The cover of ground cover types and understorey life forms was recorded using the point quadrat technique, along five randomly-selected, 20m sub-transects running perpendicular to the main transect (Figure 2, C). Life form descriptions followed the state standard for vegetation assessments (DSE 2004). These sub-transects were not in the same position as in 2002. Reference photographs were taken at the start and finish of each sub-transect. At 20cm intervals along the sub-transect each life form in contact with a perpendicular steel pin (length of 1m) was recorded, providing 100 points per sub-transect, or 500 points per site. Shrubs taller than 1m height were recorded if foliage clearly overhung the pin.
- Floristics. A comprehensive floristic search was undertaken within 10m either side of three of the randomly located sub-transects, providing three quadrats each of 400m^2. Cover-abundance was recorded using a modified Braun-Blanquet scale (Braun-Blanquet 1951). Samples of species that were not identifiable in the field were taken for later identification.
- Weeds. The cover of exotic species was recorded concurrently with the understorey structure, using the point quadrat technique. The diversity was determined in conjunction with the comprehensive floristic search.
- Canopy Species Recruitment. The diameter of every stem tree of the genus *Eucalyptus* (seedlings through to adults) within 10m each side

Table 1. Location and site details for 18 Grassy Dry Forest sites re-surveyed in 2008. The data for percentage cover (%) of exotics, bare ground and canopy are from 2002 (Tolsma and Newell, 2003) and are presented as an average (with standard deviation).

Site	Reserve date	Last burn	Easting	Northing Datum (GDA94)	Aspect	Exotics (%)	Bare Ground (%)	Canopy cover (%)
02	1997	2004	345970	5834080	SE	0.4 (0.6)	3.6 (5.1)	49.1 (11.3)
03	1997	2004	350700	5833310	NW	0 (0)	10.2 (6.4)	36.8 (6.0)
07	1975	1997	343548	5820012	SW	1.0 (1.0)	13.2 (6.5)	45.9 (6.7)
08	1975	2007	345019	5820425	NW	6.4 (7.0)	8.4 (6.1)	47.6 (19.3)
12	1997	2004	350385	5832706	W	2.2 (4.4)	20.0 (7.3)	35.5 (12.5)
14	NA	<1990	348201	5833326	W	0 (0)	26.6 (7.3)	31.0 (13.8)
17	1975	2007	345100	5820400	S	7.2 (6.9)	7.8 (5.4)	41.9 (6.4)
18	1997	<1990	350003	5833396	S	7.6 (7.6)	2.6 (1.9)	43.8 (4.0)
20	1928	<1990	354816	5838800	NW	0 (0)	30.8 (8.3)	30.8 (14.0)
25	1928	<1990	350016	5838237	E	0 (0)	8.4 (3.2)	33.0 (6.9)
28	1997	<1990	345990	5831955	NW	0 (0)	27.0 (5.3)	37.8 (10.1)
29	NA	<1990	346875	5834496	S	0 (0)	7.8 (1.6)	43.8 (22.1)
31	1997	<1990	346940	5835870	W	0 (0)	14.8 (9.9)	29.4 (4.6)
33	1997	<1990	351180	5834941	E	0 (0)	7.6 (4.8)	35.2 (5.8)
34	1997	<1990	351127	5835013	W	0 (0)	2.0 (2.3)	32.0 (12.0)
37	1928	<1990	352220	5836538	W	0 (0)	12.4 (6.5)	35.8 (8.4)
38	1928	<1990	352490	5837290	E	0 (0)	21.2 (3.3)	29.1 (2.8)
41	1928	<1990	349590	5838340	NW	0 (0)	11 (4.1)	34.8 (5.7)

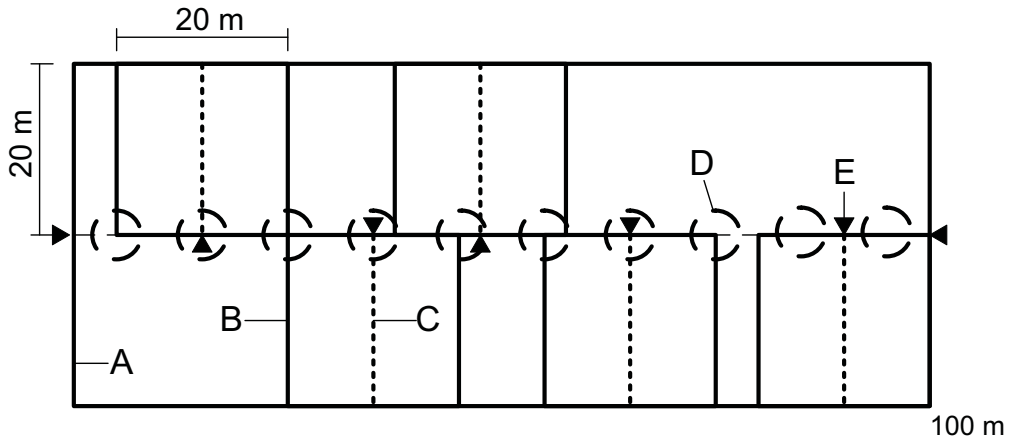


Figure 2. The basic unit of our sampling design, the 100m x 40m quadrat. The elements of the sampling include (a) the number of mature trees >50cm DBH and the length & diameter of logs (>10cm) recorded over the full quadrat; (b) 5 random 20m x 20m quadrats for the number tree stems (5), plant species diversity and cover abundance (3); (c) 5 point quadrat transects of 20m, sampled each 20cm (100 points per transect); (d) canopy photographs from tripod mounted digital camera (10m intervals); and (e) reference photographs along main and sub-transects (indicated by arrows).

of all random sub-transects was recorded. Individuals were later assigned to a size class to determine if recruitment patterns had changed over the sampling period. The density of all dead stems >10cm DBH was also recorded.

Statistical analysis

Change in vegetation layers, 2002–2008

Statistical analysis of the differences between the 2002 and 2008 data was carried out for life form (using those that occurred in greater than three sites), log length, canopy cover, floristics (native and exotic species richness and diversity), and stem density of woody species. Some life form classes were grouped due to slight differences in methodology over the two time periods. For instance, moss and rock were measured as separate classes in 2008, so were grouped with lichens and bare ground respectively. Prostrate shrubs were grouped with small shrubs because there was some debate as to which class some species should be allocated.

Most parameters were analysed separately for burnt and unburnt sites. However, tree density (including dead trees and saplings) and canopy cover were analysed together because there was little variation to suggest treatment effects.

Analyses were completed using the statistical programs R version 2.6.1 (R Development Core Team 2007) and Genstat (Lawes Agricultural Trust 2007). This study was treated as a repeated measures design, and differences were analysed using a two factor crossed ANOVA with Time and Site as factors. To analyse the changes in the length of logs an additional factor 'Size Class' (including interactions) was added. To best improve residuals, life form cover values (with the exception of medium tufted graminoids and litter) were square root transformed. Log length, density of recruits and density of dead trees were all log transformed, but species richness, tree density (>5cm DBH) and canopy cover remained untransformed.

Sample design simulations

The ability of the sampling design to detect specific changes in life form cover within a site, particularly those of low cover and/or variable cover, was also examined. Different intensities of point quadrat sampling were simulated using R version 2.6.1 (R Development Core Team 2007). The mean cover of a life form (2002 data) was expressed as a probability of detection at a given point at time one, based on 500 observations (point quadrat contacts). From 20,000 simulations we generated 95% confidence intervals for a zero change at time two for different intensities of sampling. The process was repeated with a specified % change (>0) at time two, in order to compare the level of overlap of the 95% confidence intervals. As a reference, when the proportion overlap of the 95% confidence intervals (for the >0 and 0 change scenarios) is about half, it indicates a p value of about 0.05. The point at which the lower confidence interval for the >0% change intersects with the upper confidence interval of the 0 change scenario indicates that we could be confident of the change at time two with a p value <0.01 (Cumming and Finch 2005). Power analysis was then used to determine the number of replicates that would be required to detect a certain magnitude of change between sites in a treatment group.

Results

Trees, tree recruitment and canopy cover

There were no significant changes detected in tree density (>20cm DBH) per quadrat over time (P = 0.39), with most of the change in density attributed to site variation (P = <0.001, Table 2, Figure 3). There was a strong increase in the recruitment of *Eucalyptus* juveniles (<5cm diameter) over the five year period, with a mean increase of >50% at the unburnt sites, and a 78% increase at the burnt sites (Figure 3, Table 2).

It is worth noting the number of recruits was also initially higher at the burnt sites, and largely dominated by *E. polyanthemos* (Figure 4). This may indicate there was a difference in environmental conditions at these sites. Not surprisingly, the density of recruits was dominated by the three main overstorey species; *E. macrorhyncha, E. polyanthemos* and *E. goniocalyx*. The magnitude of the increase in recruits seemed to be fairly consistent across species, and proportional to the number of recruits in 2002 at all sites (Figure 4).

The average density of dead stems (>10cm DBH) also increased seven-fold over the five year time period, by approximately 1.8 trees per quadrat on average (Figure 3, Table 2). Of the dead trees recorded across the study sites in 2008, 41% were *E. goniocalyx*, 18% were *E. macrorhyncha*, 9% were *E. polyanthemos* and 24% could not be identified. The distribution of mortality by stem size classes was not distinct from the living stem data, suggesting tree death was not disproportionately associated with the any size class.

No difference was found in the average change in density of dead trees over time between the unburnt and burnt quadrats (mean difference ~ 0.8 trees/quadrat, 95% CI = 1.1 trees). Canopy cover did not change significantly over the five year period, with a mean difference of +1.5% (95% CI = 2.5 %) between 2002 and 2008 (Table 2, Figure 3).

Table 2. Two factor crossed ANOVA for log cover, density of recruits and trees (including dead trees), and canopy cover at unburnt (18) and burnt sites (5).

Attribute	Source of variation (and d.f: unburnt, burnt)	Unburnt Sites			Burnt sites		
		F	P	MS residual	F	P	MS residual
Logs	Time (1,1)	15.62	<0.001	0.70	29.73	<0.001	0.6
	Site (12, 4)	7.05	<0.001		2.27	0.11	
	Size (4, 4)	62.91	<0.001		24.94	<0.001	
	Time:Site (12, 4)	1.17	0.33		1.58	0.23	
	Time:Size (4, 4)	1.12	0.36		6.74	<0.01	
	Site:Size (48, 16)	1.64	<0.05		0.97	0.52	
	Residuals (48, 16)						
Recruits	Time (1, 1)	8.13	<0.01	0.06	5.43	<0.05	0.06
(<5cm, DBH)	Site (12, 4)	11.36	<0.001		5.22	<0.01	
	Time:Site (12, 4)	1.24	0.26		1.62	0.19	
	Residuals (104, 40)						
Tree density	Time (1)	0.74	0.39	12.03	NA		
(>20cm, DBH)	Site (17)	5.06	<0.001				
	Time:Site (17)	1.04	0.42				
	Residuals (144)						
Tree density	Time (1)	0.01	0.94	63.23	NA		
(>5 - <20cm,	Site (17)	13.68	<0.001				
DBH)	Time:Site (17)	1.04	0.42				
	Residuals (144)						
Dead trees	Time (1)	95.09	<0.001	0.05	NA		
	Site (4)	1.44	0.13				
	Time:Site (4)	1.23	0.25				
	Residuals (40)						
Canopy Cover	Time (1)	1.80	0.18	112.00	NA		
	Site (17)	5.76	<0.001				
	Time:Site (17)	3.25	<0.001				
	Residuals (324)						

Ground, understorey and mid-storey cover

Bare ground cover did not change significantly at the unburnt sites, whilst at the burnt sites bare ground cover increased markedly over the same period from around 10% to over 30% (Figure 6, Table 3). Conversely, litter cover almost doubled in unburnt sites ($p<0.001$) whereas there was no significant net change in litter cover at the burnt sites (Figure 6, Table 3). The cover of the smallest diameter logs (10–20cm) – accounting for the majority of log cover – also significantly increased (mean increase >30%) in sites that were not burnt (Figure 6, Table 3). However, there was no pattern of increase considering all log diameter classes together (Table 2).

Soil crust cover was generally low across all sites, both 2002 and 2008 (mean = 0.7%). Nonetheless, there was a significant four-fold increase in soil crust cover over the five year period at the unburnt sites (Table 3, Figure 6). By comparison, at the burnt sites there was a small decline in cover over time, which nonetheless constituted a decline of almost 90% against 2002 levels.

There was a low cover (mean cover 4.4%, s.e. 0.5%) of bryophytes and lichen at the unburnt sites, which was slightly higher at the second sampling (Figure 6). Cover at the burnt sites was initially higher on average, and highly variable, but dropped significantly by 75% to a far less variable cover in 2008 (Table 3). In both cases, there was a high level of variation in cover associated with site differences.

There was a small decline in cover of medium tufted graminoids (MTG) at the unburnt sites (Figure 6, Table 3), which was equivalent to an 8.3% decrease in relation to the initial cover (95% CI = 15.6%). Most of the variation in the cover of MTG could be attributed to site differences (Table

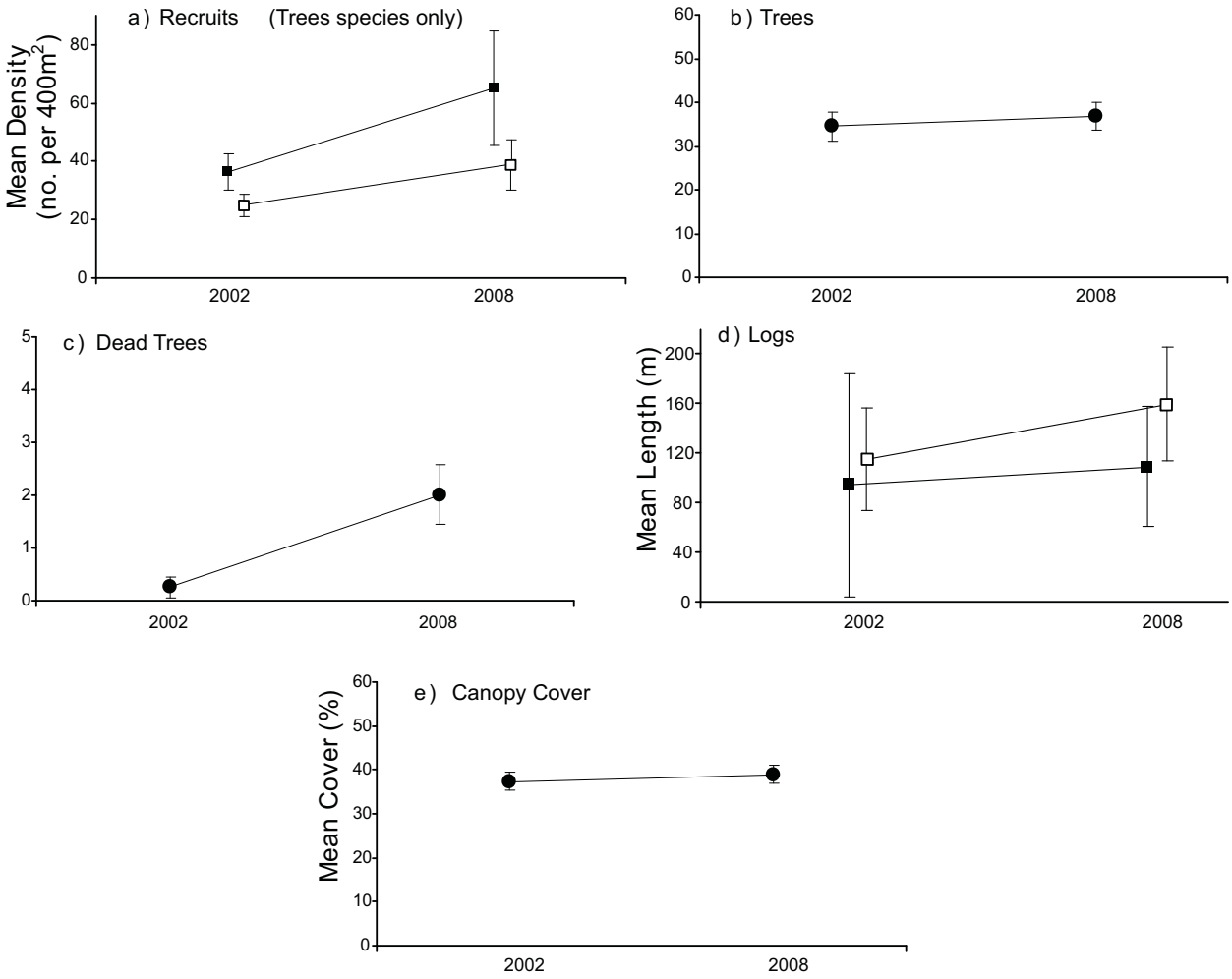


Figure 3. Changes over the time period from 2002–2008 at unburnt (open squares; n = 65 for recruits, n = 13 for logs), burnt (filled squares; n = 25 for recruits, n = 5 for logs) and all (filled circles; n = 90 for trees/dead trees, n = 180 for canopy cover) sites in the a) mean density of recruits (<5cm DBH), b) mean density of trees (>20cm DBH), c) mean density of dead trees (>10cm, DBH), d) mean length (in metres) of logs, and e) mean canopy cover (%). All graphs show 95% confidence interval bars.

3). At the burnt sites, the magnitude of the average decrease over time in relation to the 2002 cover was greater (42.1%, 95% CI = 16.8%; Figure 6, Table 3).

The average cover of medium shrubs was low and highly variable (Figure 6) in both 2002 and 2008, with no obvious change in cover over time (difference in means = 0.1%, 95% CI = 0.9%). The cover of small shrubs was similarly low at all sites in 2002 and 2008 (Figure 6). At the unburnt sites, there was no conclusive change in cover over time (difference in means = 0.4%, 95% CI = 1.0%). However, at the burnt sites, the average 37.5% decrease in shrub cover was significant despite large variation around the mean in 2008 (95% CI = 65.9%; Figure 6, Table 3).

The initial herb cover at all sites was very low (average cover <2%), but in general declined consistently across sites over time. All sites recorded a significant ($p<0.001$) decrease in the cover of medium herbs over time (Figure 6, Table 3), with an 86.1% decline (95% CI = 45.5%) at the unburnt sites and a 71.7% decline (95% CI = 45.3%) at the burnt sites.

The cover of small herbs did not change significantly at the unburnt sites over time (Figure 6), but the average 80.8% drop in cover (95% CI = 67.4%) at the burnt sites was consistent (Table 3). Figure 8 indicates that the loss in cover of herbs corresponded to a loss in herb species richness, particularly in relation to small herbs.

There was a drop in the cover of exotics over time, with weed species detected in 33.3% of the transect points in 2002, and no weeds were detected in the point quadrats at any sites in 2008.

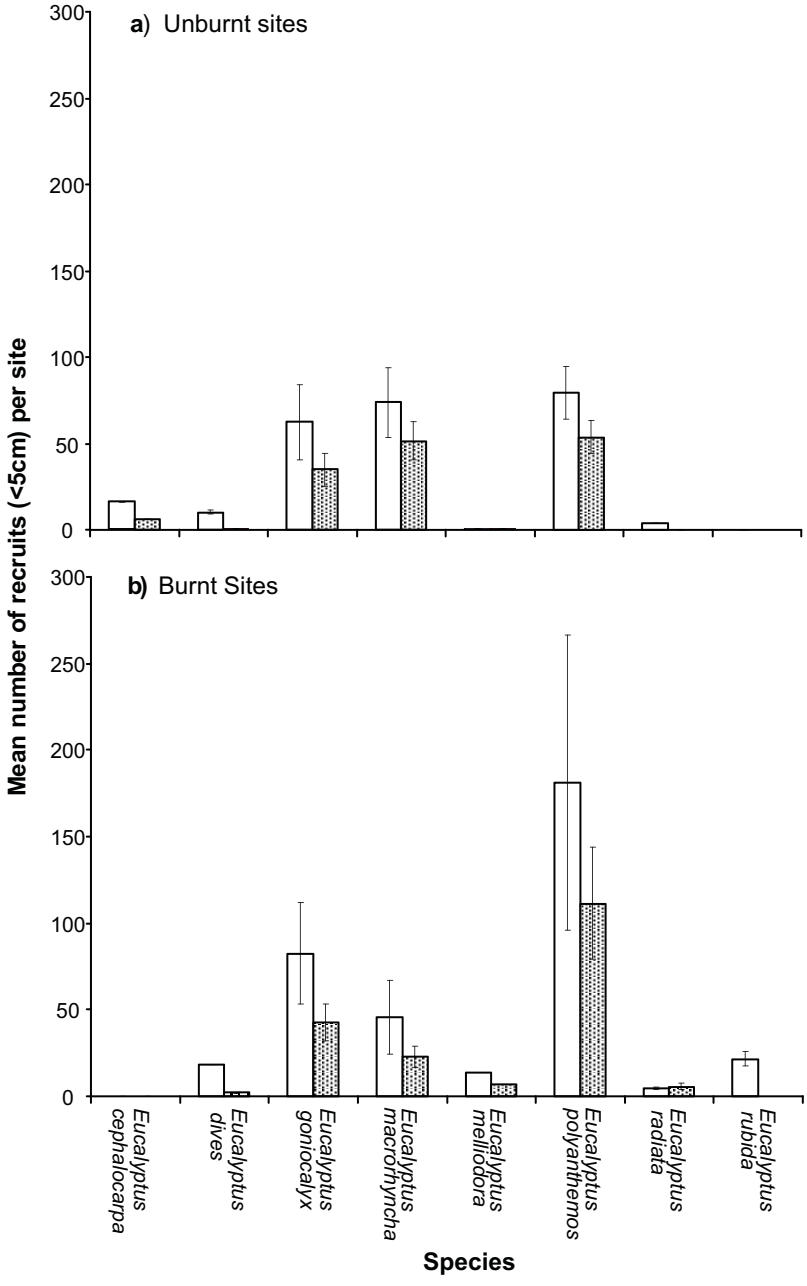


Figure 4.
Mean number of recruits (per 4000m^2 quadrat) in 2002 (white bars) and 2008 (dotted bars) in relation to species, with standard error bars

Floristic changes

Native species richness decreased consistently across sites over the sampling period by 12% on the unburnt sites and by 28% on the burnt sites (Table 4, Figure 7). Weed species richness was much lower across sites, but the magnitude of change over the time period was greater, dropping by 72% and 69% on the unburnt and burnt sites respectively (Figure 7, Table 4). Species richness (of weeds and natives) was initially higher at the burnt sites, but was similar across all sites in 2008. On examination of the average difference in the number of species grouped according to life form over the time period (Figure 8), it is evident that the decrease in richness was predominantly related to a loss of herbaceous species, particularly medium herbs. Along with the decrease in species richness, there was a consistent increase in species evenness on unburnt and burnt sites over time. This is an indication that the relative abundance of individuals was more evenly

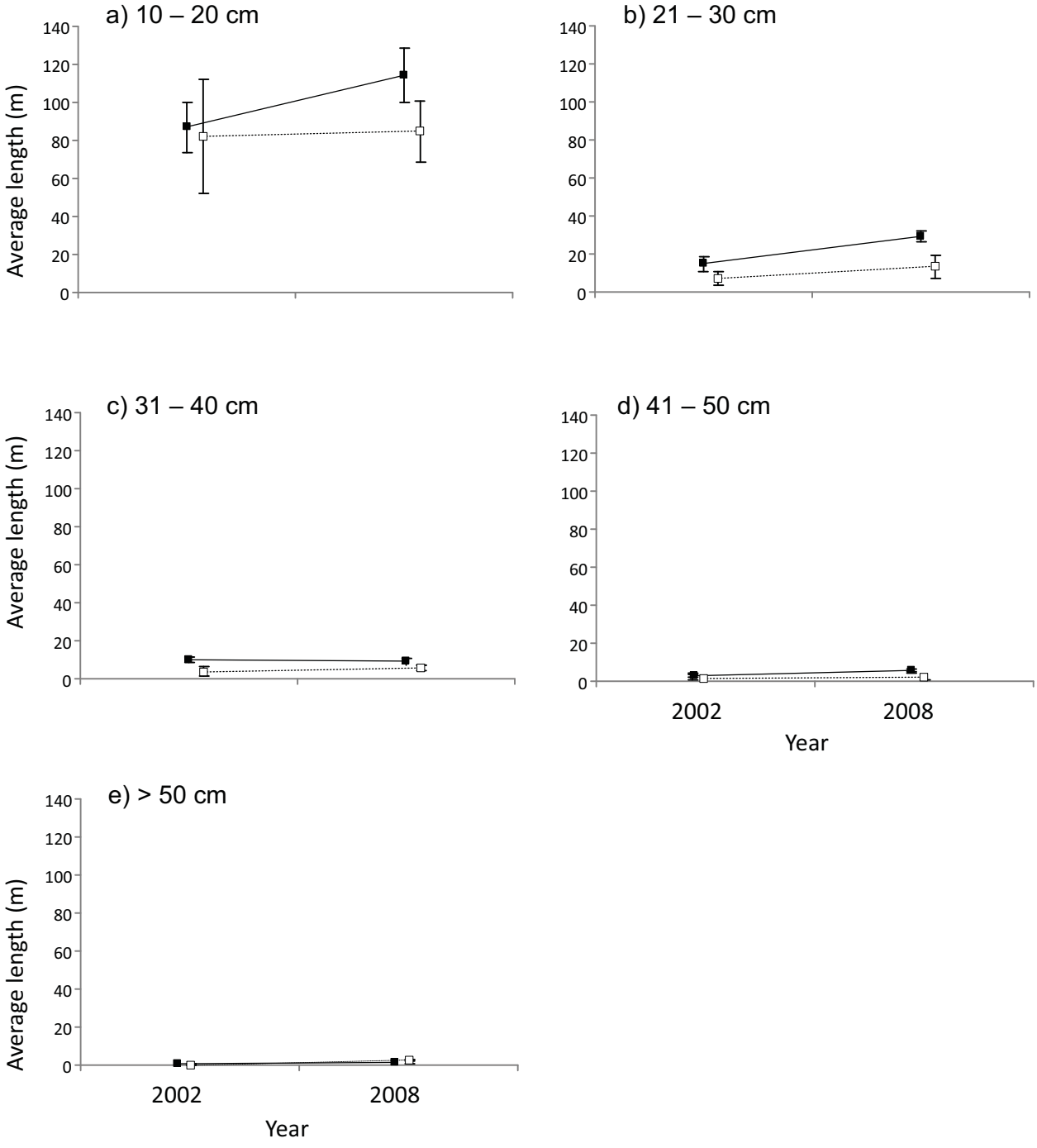


Figure 5. Mean difference in total length of logs (per 4000 m² quadrat) from 2002–2008 in relation to size class, with 95% confidence interval bars. The filled squares represent the burnt sites (n = 5 sites) and the open squares represent the unburnt sites (n = 13 sites).

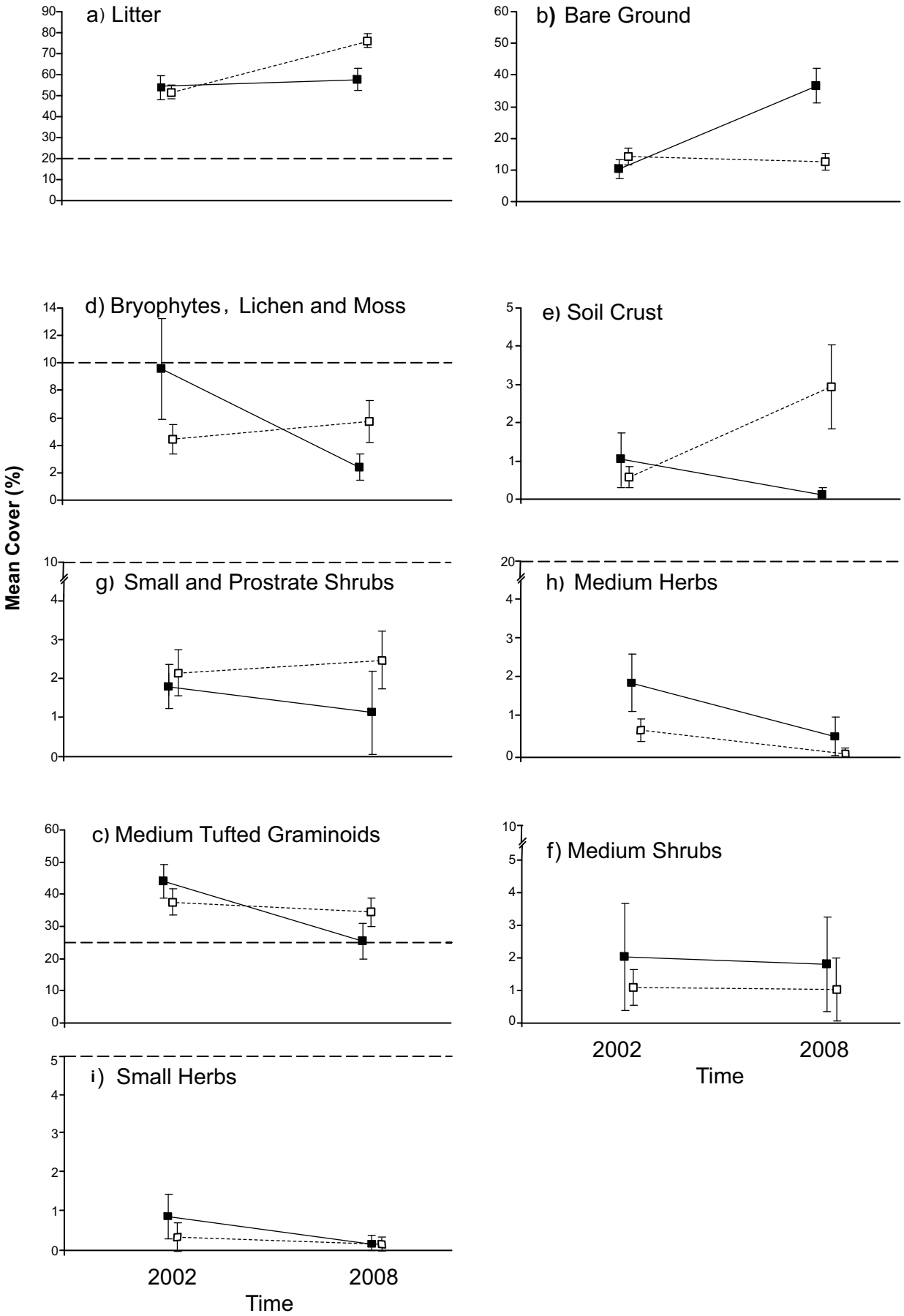


Figure 6. Mean percentage cover (± 95% confidence intervals) of life form classes in 2002 and 2008 at burnt (filled symbol, n = 25 quadrats) and unburnt sites (open symbol, n = 65 quadrats). Dashed horizontal reference lines indicate the benchmark cover for structure class and life forms in Grassy Dry Forest (EVC 22) in the Highlands – Southern Fall bioregion.

Table 3. Two factor crossed ANOVA for the cover of vegetative life forms, bare ground and litter. The degrees of freedom (unburnt sites, burnt sites) are as follows: Time (1, 1), Site (12, 4), Time:Site (12, 4) and Residuals (104, 40).

Life form	Source of variation	Unburnt Sites			Burnt sites		
		F	P values	MS residual	F	P values	MS residual
Litter	Time	164.03	<0.001	120.00	0.86	0.36	131.0
	Site	3.58	<0.001		3.66	<0.05	
	Time:Site	4.09	<0.001		0.89	0.48	
Bare Ground	Time	1.53	0.22	0.89	124.02	<0.001	0.90
	Site	15.46	<0.001		5.09	<0.01	
	Time:Site	1.92	<0.05		2.34	0.07	
Soil Crust	Time	32.94	<0.001	0.50	7.84	<0.01	0.39
	Site	7.51	<0.001		1.87	0.13	
	Time:Site	5.65	<0.001		0.88	0.48	
Bryophytes, Lichens & Moss	Time	1.86	0.175	1.00	31.27	<0.001	0.80
	Site	7.31	<0.001		10.72	<0.001	
	Time:Site	1.73	0.071		2.65	<0.05	
Medium Tufted Graminoids	Time	3.63	0.06	87.00	39.24	<0.001	110.0
	Site	25.27	<0.001		6.78	<0.001	
	Time:Site	1.16	0.32		1.82	0.144	
Medium Shrubs	Time	1.55	0.22	0.60	0.07	0.79	1.00
	Site	1.48	0.14		1.12	0.36	
	Time:Site	2.44	<0.01		2.91	<0.05	
Small and prostrate shrubs	Time	0..02	0.90	0.60	10.19	<0.01	0.53
	Site	7.01	<0.001		0.65	0.63	
	Time:Site	1.48	0.14		3.67	<0.05	
Medium herbs	Time	25.87	<0.001	0.26	17.36	<0.001	0.49
	Site	2.00	<0.05		0.81	0.52	
	Time:Site	1.87	<0.05		3.16	0.24	
Small herbs	Time	1.52	0.22	0.22	9.46	<0.01	0.30
	Site	3.39	<0.001		2.99	<0.05	
	Time:Site	1.94	<0.05		4.50	<0.01	

Table 4. Two factor crossed ANOVA for species richness and diversity at unburnt (18) and burnt sites (5).

Attribute	Source of variation (and d.f: unburnt, burnt)	Unburnt sites (n = 13)			Burnt sites (n = 5)		
		F	P values	MS residual	F	P values	MS residual
Species Richness (natives)	Time (1,1)	14.75	<0.001	13.79	47.21	<0.001	15.47
	Site (12,4)	8.47	<0.001		17.87	<0.001	
	Time:Site (12, 4)	1.19	0.32		1.36	0.29	
	Residuals (52, 20)						
Species Richness (weeds)	Time (1,1)	29.91	<0.001	0.03	22.30	<0.001	0.03
	Site (12,4)	7.89	<0.001		7.84	<0.001	
	Time:Site (12, 4)	2.54	<0.05		3.58	0.02	
	Residuals (52, 20)						
Eveness (all)	Time (1,1)	153.34	<0.001	0.10	23.64	<0.001	0.19
	Site (12,4)	13.70	<0.001		9.06	<0.001	
	Time:Site (12,4)	7.98	<0.001		1.87	0.16	
	Residuals (52, 20)						

distributed among the species present in quadrats in 2008 (Table 4, Figure 7).

Sample size simulations for detecting change within a site

Detecting change in a life form with low cover (soil crust)

This simulation used sampling intensity applied in this field study as its starting point. Thus, the sampling intensity at time A is set at 500 point quadrat hits. Figure 9 illustrates the sample size required in a second sampling period to detect various magnitudes of change with 95% confidence in the cover of soil crust at one site based on a mean at time A of 0.06%. The detection of a three-fold change (i.e. from 0.06 to 0.18%) over time would be unlikely without increasing the sampling effort (Figure 9a). For instance, the point at which the proportion overlap in confidence intervals is no more than a half (Cumming and Finch 2005) occurs at approximately at 1000 point quadrat hits at time B (Figure 9a). A sampling intensity of just over 1500 points at time B (or roughly 3 times the initial sampling intensity) would be required before the 95% confidence intervals do not overlap, equivalent to $p<0.01$.

Detecting a four-fold change (i.e. from 0.06 to 0.24%, Figure 9b) with a p value of about 0.05 would require the sample size to be increased to about 750 points (1.5 times the initial effort) at time two, whereas a doubling of the sampling intensity at time B (i.e. 1000 points) would be required to be confident of detecting the change with a p value <0.01. We could be 95% confident of detecting a five-fold change (i.e. from 0.06 to 0.3%, Figure 9c) with $P<0.01$ without any increase in sampling intensity.

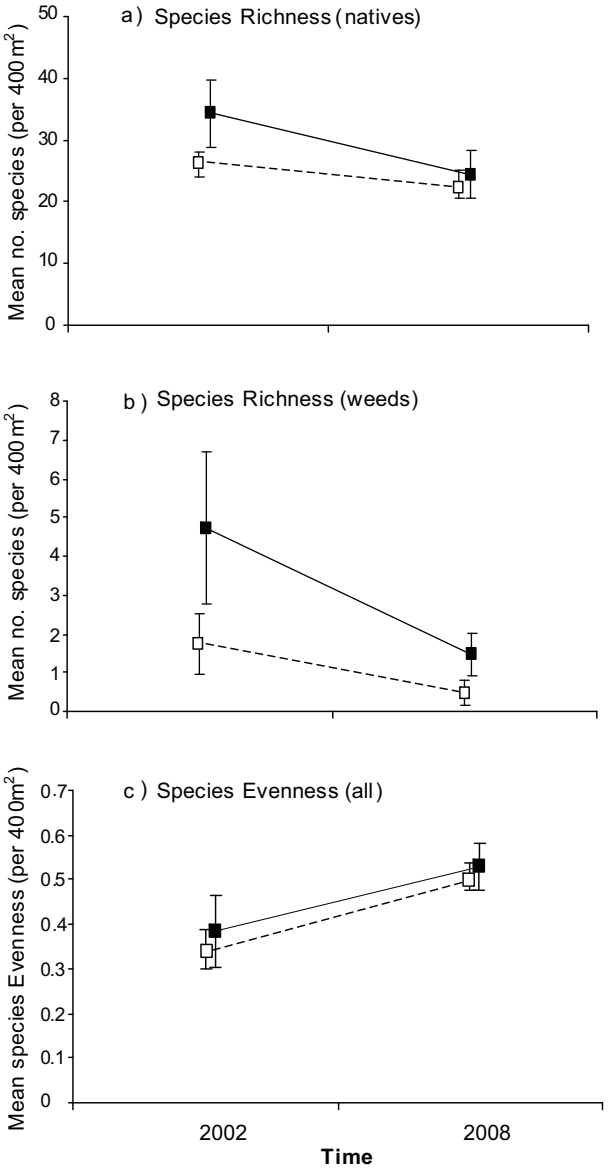


Figure 7. Changes in mean species richness (per 400m² quadrat) with 95% confidence interval bars of natives (a) and exotics (b), and overall species diversity (c) over the time period from 2002–2008 at unburnt (open squares) and burnt (filled squares) sites.

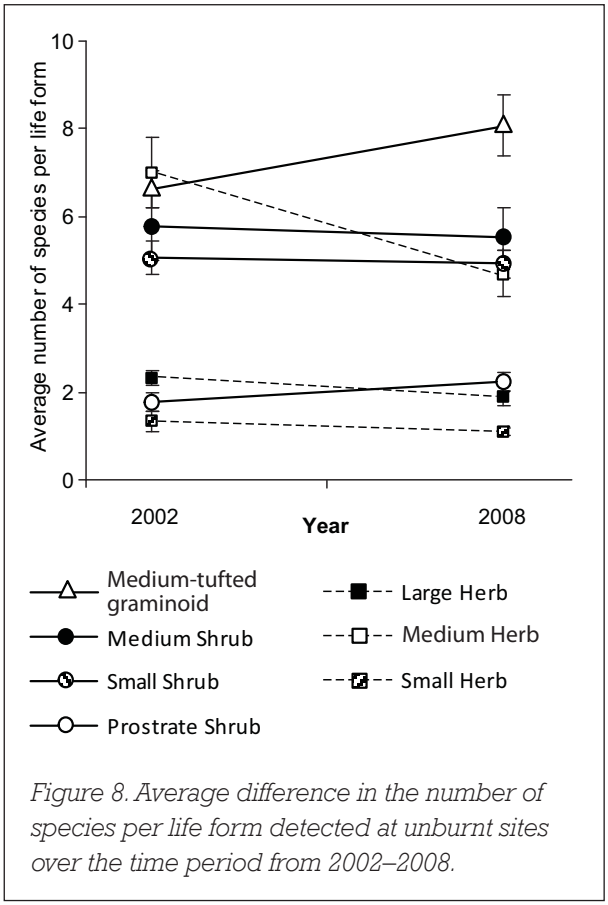


Figure 8. Average difference in the number of species per life form detected at unburnt sites over the time period from 2002–2008.

Figure 9.
Changes in the mean cover of soil crust (with 95% confidence bounds) with different sampling intensities. In each figure the black lines represent a zero change in cover from 0.06% at time B.
a) change in cover from 0.06% to 0.18%,
b) change in cover from 0.06% to 0.24%, and
c) change in cover from 0.06% to 0.3%.

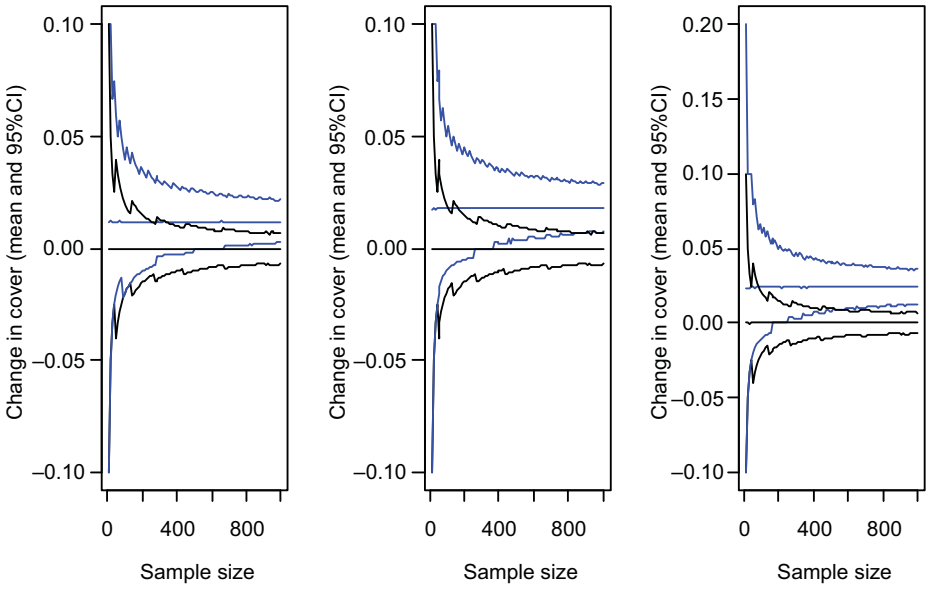


Figure 10.
Changes in the mean cover of medium tufted graminoids (with 95% confidence bounds) with different sampling intensities. In each figure, the black lines represent a zero change in cover from 37.6% at time B. The scenarios were for increases of: a) 10%, b) 20% and c) 30%.

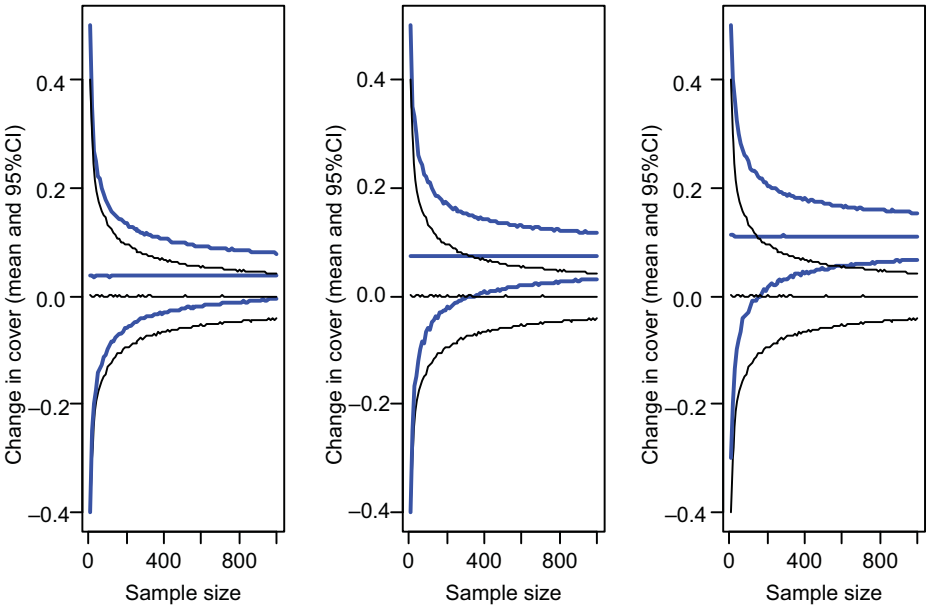
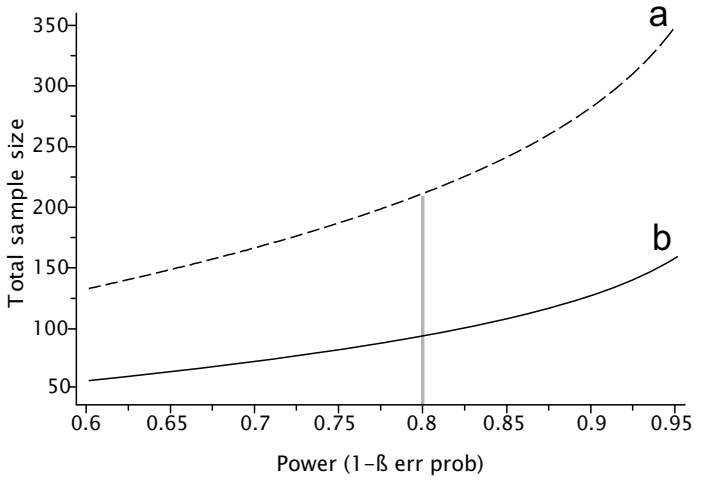


Figure 11.
Power curves (power vs. sample size) illustrating the number of quadrats required to detect:
a) a 10% increase (effect size dz = 0.19) and
b) a 15% increase (effect size dz = 0.29) in the cover of medium tufted graminoids in a second sampling period.
Test is for difference between two dependant means (matched pairs).
Two tailed, α err prob = 0.05.



Detecting change in a life form with medium-high cover (medium tufted graminoids)

The calculations for Figure 10 are based on the cover of medium tufted graminoids, which recorded a medium to high cover over the sampling period. In this instance the initial mean cover was 37.6% and we were attempting to detect 10–30% changes in cover with 95% confidence at time B.

It would be unlikely to detect a 10% change in cover (37.6–41.4%, Figure 10a) even with a dramatic increase in sampling intensity. However, a 20% change (37.6–45.1%, Figure 10b) would only require the sampling intensity to increase to about 800 points (or 1.6 times the original sampling intensity) to be 95% confident of the change with $p = 0.05$, or 1300 points (or 2.6 times the original sampling intensity) to be confident with $p<0.01$. A 30% change (37.6–48.9%, Figure 10c) would not require a change in sampling intensity in the second sampling period to be 95% confident of detecting the change with $p<0.01$.

Simulations for detecting change between sites

The next step was to determine what replication at the site level would be required to detect an underlying change of a certain magnitude between sites. For this we based our calculations on the average cover of medium tufted graminoids in the 2008 sampling period. The mean cover was 34.5% (sd 15.4), which was derived from 13 sites, with 5 transects per site (65 transects in total). Assuming a constant standard deviation over the sampling period, to detect a significant 10% change in cover (i.e. +3.5) with a power threshold of 0.8 would require a 3.2 fold increase in sampling effort (i.e. 211 transects; Figure 11a), whereas to detect a significant 15% change in the cover of medium tufted graminoids (i.e. +5.9) with a power of 0.8 would require 95 transects, which is a 150% increase in sampling effort (Figure 11b).

Discussion

Compositional and structural change

Our first aim was to determine whether change in vegetation composition and structure could be detected in a maturing dry forest system after a five-year period. We found that consistent changes in key structural and compositional variables could be detected.

The typical pattern in unburnt sites was an accumulation of forest floor litter and small diameter logs, ongoing recruitment and mortality amongst the canopy eucalypts, and a decrease in cover of sensitive life forms such as herbs. The cover of bare ground, graminoids and shrubs did not change significantly. Sites that were burnt in the intervening period exhibited no net change in litter and log cover, an increase in bare ground cover, and strong decreases in the cover of biological soil crust. Burnt sites also had a significant decrease in biomass of graminoids, shrubs and herbs compared with their 2002 levels.

Changes in land use associated with European settlement are placing increasing pressure on these vegetation communities. Over the past century, these forests have been subject to logging, increasing fragmentation as a result of development at the urban fringe, invasions by weeds and feral animals, grazing by domestic livestock and native animals, ongoing periods of drought, and climate change. Additionally, an altered fire regime is generally experienced in areas of native vegetation close to urban settlement (Gill and Williams 1996; Kirkpatrick 2004). With respect to changes occurring in the last five years, two likely factors were identified as drivers of vegetative change in these systems; succession in the absence of fire (i.e. through the suppression of wildfires and low incidence of prescribed burning), and drought.

Disturbance such as fire is a primary driver of forest structure and function in many Australian forest and woodland systems (Attiwill 1994; Yates and Hobbs 1997; Hobbs 2002; Wardle et al. 2004; Jurskis 2005). In addition, long-term rainfall trends can facilitate sustained changes in vegetation structure and composition (Fensham and Holman 1999; Fensham et al. 2005). It is likely that these factors were acting in conjunction to shape forest structure and composition, and distinguishing between their influences is not possible without further examination of sites over an extended period of time, with variable fire management and rainfall conditions.

Under benign conditions, it was likely that only small changes in the structure of the overstorey component would occur over a five year period in the absence of major fire at these sites. However, changes might have occurred as a result of the long-running sequence of below average rainfall years (Bureau of Meteorology data). Drought associated plant mortality is a worldwide phenomena (McDowell et al. 2008), and has been the proposed cause of dieback and decline in other Eucalyptus forests and woodlands in Australia (Pook 1980; Landsberg 1985; Fensham and Holman 1999; Rice et al. 2004; Fensham et al. 2005; Davidson et al. 2007; Fensham and Fairfax 2007). This can occur directly via hydraulic failure and possible carbon starvation as a result of stomatal closure (McDowell et al. 2008).

There was no significant change in canopy cover or tree density, even though the total number of dead trees increased almost eight-fold over the entire study area. Whilst tree fall and loss of older (senescence) or younger individuals (suppression and self thinning) is a natural feature of forest development, this increase in mortality was distributed throughout the size classes and may have been due to drought conditions. Of the three dominant *Eucalyptus* species (*E. macrorhyncha*, *E. polyanthemos* and *E. goniocalyx*), *E. goniocalyx* recorded the highest mortality rates in both years, though a large proportion of dead trees in 2008 were unable to be identified. Previous records of dieback in these species in drought affected dry sclerophyll forest by Pook (1980) found *E. macrorhyncha* to be the most susceptible to death, and *E. goniocalyx* the most susceptible to a severe decline in crown health.

In principle, a reduction in canopy cover might be expected with increased tree mortality. The large increase in both the cover of fine (predominantly Eucalyptus) litter and coarse woody debris in the five year period at the unburnt sites indicates loss was occurring in the canopy layer. It could be that there was no net change (i.e., replacement), or that the sampling resolution of the canopy cover estimates was too coarse. At burnt sites, the cover of litter and coarse woody debris was maintained at 2002 levels. High litter accumulation rates are a feature of many Australian forest systems (Attiwill and Leeper, 1987), and are known to increase as forests age (Mackey and Smail 1995).

Recruitment of *Eucalyptus* juveniles (<5cm diameter) increased over the sampling period at both unburnt and burnt sites. Whilst we believe that the recorded increase was largely due to the establishment of new individuals, the total change includes some resprouting from the base of established individuals (coppicing). The number of stems increased at both unburnt and burnt sites, indicating fire was not the sole cause of recruitment. There was no evidence to suggest an overall decline in tree density at

these sites, as tree density and recruitment did not decrease over the period irrespective of increased mortality. As such, it appeared that drought was not limiting recruitment or growth.

There were consistent structural and compositional changes in the understorey in the absence of fire. For instance, species richness of natives and weeds was significantly lower in 2008, though the magnitude of change was much greater for the weed species. Accompanying this was a significant increase in the cover of litter and soil crust, and a reduction in cover of medium herbs. Though the latter two life forms were already low in cover, the magnitude of the change over time was large (i.e. up to four-fold for soil crust). When species data from quadrats was classified by life form groups, we found a decrease in herb species richness to accompany the decrease in cover of herbs from point quadrat data.

Not surprisingly, life form cover and floristic changes at the burned sites tended to be of greater magnitude than the changes at unburned sites. There was a significant increase in the cover of bare ground correlated with a decrease in cover of medium tufted graminoids, indicating grassy cover had been replaced by bare ground. The cover of bryophytes and lichen, soil crust, small and prostrate shrubs, medium herbs, small herbs, and species richness also decreased markedly in burned sites. These results suggest that the incidence of a low intensity prescribed burn has the capability to cause significant compositional and structural changes in the understorey in the short-term. Further monitoring could elucidate the potential impacts of fuel reduction burning on successional changes in understorey composition and structure.

Adequacy of the sampling design

Our second aim was to determine whether the original sampling design and intensity was adequate to detect specific changes in the cover of life form classes within sites, particularly those of low and/or variable cover. It seems prudent to design a monitoring system around detection of small changes, not just those that ought to trigger management response. For this reason our point quadrat sampling intensity simulations explored not only what magnitude of change our sampling was able to detect, but also what intensity might have been required to detect more subtle changes. On an individual site basis, the sampling intensity used in this study was unlikely to detect a three-fold increase from a starting cover of less than 1%, or a 10% change from a starting cover of around 30% (i.e. to 33%). However, because we had multiple site replicates, and observed changes were fairly consistent amongst replicates, our field study did find significant differences for some small magnitude changes. For example, the initial cover of medium herbs was <2%, but we were able to detect a highly significant – though small magnitude – decrease in cover over the five year period.

In order to detect small changes in cover over time within a single site with the point quadrat technique we would have to have sampled more than 500 points at time one. Given that the greater part of the time-cost involved in the point quadrat method is in setting up transects, rather than actual sampling, an increase in the initial sampling effort to 1000 points per site might be feasible.

We also examined the number of transect replicates that might be required to adequately detect a given change in cover across the study area. Power analyses suggested that our method was sufficient to detect some of our observed changes, but hundreds more transects may be required to be confident of detecting the smallest changes. Considering the time taken to sample one site using the current sampling design, increasing the number of site replicates is likely to be more time-consuming and costly than increasing survey effort within a site. It has to be remembered that if increased replication is not feasible, one could elect to set the acceptable type I (false positive) error rate higher than the arbitrary convention of 5% (Foster 2001; Di Stefano 2003).

The question of where to draw the line in requisite sensitivity of monitoring for native vegetation condition change has been summarised as a matter of distinguishing between changes of (ecological) interest, versus changes of (management) concern (DSE 2008). However, this appealing framework has a paradox at its heart. Unless we characterise natural variation sufficiently, we may never be confident of the distinction between these types of change. For example, it is quite possible that managers, scientists, and the community might agree that in most cases gearing up a monitoring program to detect an increase in mean cover from 0.6% to 1.8% in the cover of native herbs over a five-year time step would be excessive. However, if the direction of change were reversed, a decrease over five years from 1.8% to 0.6% cover, perhaps some alarm might be raised. A repeat over a subsequent five year monitoring period could see the cover reach 0% and depending on the species and the system in question, that could be irreversible. These simulations provide valuable input into an overdue debate about adaptive management of native vegetation. As this study has illustrated, the missing ingredients are agreed statements about acceptable thresholds of change, certainty about change, and timeliness of the awareness of change in native vegetation.

Conclusion

This study detected changes in structure and composition in dry grassy forest that could be plausibly linked to the effects of drought and fire. However it was not possible to confirm whether these changes were directional or normal cyclical fluctuations. A distinction between ''changes of interest'' and ''changes of concern'' has been proposed, which serves to highlight that it is not desirable or possible for land managers to micro-manage nature in all its variability. Whilst the variation within natural systems is of interest to ecologists, land managers need to know when undesirable directional changes (changes of concern) occur that may require intervention, such as weed incursions or sustained declines in species or functional groups. Unless the nature and magnitude of inherent variability is understood, we cannot expect to be able to identify changes of concern early enough to successfully intervene.

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