Chapter 9: Fire and Granitic Outcrops

Fire trial plots near the border track in Bald Rock National Park

Chapter 9

Fire and Granitic Outcrops

9.1 Introduction

Fire as a disturbance is an important factor structuring many plant communities and has the potential to influence the occurrence and density of the species in a community. Fires are known to affect vegetation directly and indirectly, by killing some organs or plants and stimulating others (Williams & Gill 1995). Within the area of the New England Batholith, particularly in the east, fires are a major controlling factor in determining composition, richness and diversity. Section 3.4 alluded to the effect of fire on increasing the effective insularity of outcrop flora in some regions of the Batholith, by inducing the germination of fire-ephemeral species (Binns 1995a). Fire is also known to decrease the effective insularity of outcrops by decreasing the number of outcrop restricted taxa and increasing ubiquitous species (Binns 1992). Most of the literature regarding fire and the responses of flora to fire is based on forests that are valuable for timber (Boughton 1970), or on areas with close proximity to major populated areas, and therefore is not closely relevant to granitic outcrop floras.

The communities on the eastern portion of the New England Batholith form heathland and shrubland formations (Chapter 4). Most of the fire studies of heath vegetation in Australia have been carried out in coastal south eastern Australia (Williams 1995). It is commonly thought that all sclerophyllous vegetation of eastern Australia is fundamentally affected by fire for example, Specht (1981) states that 'In general, dense sclerophyllous vegetation tends to contain many species with remarkable powers of regeneration following fire ...'. He then further states that 'Of all the factors involved in seedling germination and subsequent survival in heathland areas, fire appears as a major determining factor ... management programs for heathland areas should place due emphasis on the role of fire in seedling germination'. Gill and Bradstock (1981) considered that fire was of major importance to the perpetuation of heathlands of the world.

The floras on granitic outcrops in Australia contain species that are congeneric with those contained within the surrounding flora. This broad floristic and structural similarity (Chapter 4) has led to vegetation on granitic outcrops of the New England Batholith being assigned a frequent fire regime that is the same as structurally similar communities (often 10—15 years) in National Parks and Wildlife Fire Management documents (Hunter 1998a; Hunter 1999a). This is despite many publications that indicate the flora on outcrops is not fire adapted (Gillham 1961; Erickson *et al.* 1973; Ashton & Webb 1977; Craven & Jones 1991; Fuls *et al.* 1992; Binns 1995a, b; Gröger & Barthlott 1996; Beard 1997; Hopper *et al.* 1997; Heinze *et al.* 1998; Hunter 1998a; Lawler *et al.* 1998; Hopper 1999). Fires of this frequency are likely to decrease the richness and diversity of these systems.

However, no quantitative information currently exists about fire responses of the flora and vegetation communities on granitic outcrops. It is unknown what the basic responses of individual species or communities are. In particular, how fire may effect the composition and thereby the richness and diversity of a system is unknown. The insular nature of outcrops and their component floras makes it likely that a large-scale disturbance, such as fire, may cause extinction of many populations due to their small size. Yet it may also permit many populations to remain unaffected, as the system is not contiguous, so all parts may not be simultaneously affected. Basic information on the responses of individual species to fire and its effects on composition, richness and diversity is needed.

The aim of this study is to examine compositional, diversity and richness changes of granitic outcrop vegetation after fire. Specific questions asked include:

- Is there a difference between the responses of the outcrop flora and the surrounding forest flora to fire?
- Does the composition of the flora on outcrops approach that of the forests after fire or after repeated fires?
- Are changes induced by fire due solely to the decrease in competition and increase in available light?
- Does a fire-ephemeral flora exist on granitic outcrops?

9.2 Methods

9.2.1 Opportunistic observations and qualitative autecological methodology

During the survey period many large and intensive fires occurred during November 1994 and February 1995 throughout the eastern portion of the study region (*pers. obs.*). This enabled fire responses to be recorded for taxa and some autecological information to be gathered. Qualitative information on changes in structure and floristics after fire was gathered at Bald Rock and Girraween National Parks (BR), Butterleaf (BL) and Backwater (BC) (Figure 2.1). Populations of *Muehlenbeckia costata* and *Monotaxis macrophylla* were followed across their range for two years between 1994 and 1996. Although no quadrats were formally placed prior to fire, sites had been visited prior to wildfire in planning stratification methods and inspection of access trails, and notes were taken during these visits on dominant species on outcrops. These notes, and the changes recorded subsequently, constitute important qualitative information on changes in structure and floristics after fire.

9.2.2 Seed germination experiments

During the survey two herbaceous, potentially fire responsive species *Muehlenbeckia costata* and *Monotaxis macrophylla* dominate granitic outcrops after the passage of recent fires. Both taxa are considered threatened within the study region (*Threatened Species Conservation Act, NSW Government 1998*). In order to discover more about the fire responses of these two significant species, germination trials were conducted.

Muehlenbeckia costata (Polygonaceae) is dioecious, with stems procumbent and weakly climbing. The stems may reach up to 5 m in length radiating from a central rootstock. The leaves are ovate, oblong to almost triangular, 3-14 cm long, 1-9 cm wide with a cordate base and crenulate margins. The nuts are trigonous with a hard, black and rugose coat. The perianth is initially green, but as the nut matures it becomes orange and fleshy and elongates and swells to enclose the nut, becoming the diaspore.

Monotaxis macrophylla (Euphorbiaceae) is a glabrous herb to about 50 cm tall with leaves 2-5 cm long and 0.2-2.5 cm wide which are thin and entire or toothed. The plants are monoecious and the flowers are yellow and form dense terminal clusters.

Diaspores were collected from taxa in the field and each subjected to 24 separate treatments. Heat treatments were; no treatment, 60°C, 80°C, 100°C and 120°C (each for ten minutes). Other treatments included smoking with native fuel, scarification via sandpaper, fungicidal treatment with Benlate 0.5g/l, phosphate buffering (pH 4.2) and the associate controls. Germination trays were randomized in incubators. All diaspores were incubated on a 12 hr day and 12 hr night light cycle and temperature variation of 20°C at night and 30°C by day. Numbers of germinated seeds were checked once a week for 8 weeks.

Muehlenbeckia costata seeds were collected from mature plants with additional small amounts of seed gained from neighboring soil. A total of 25 *M. costata* seeds were placed in each treatment tray (600 seeds) with an additional 200 seeds used for viability tests. All *Monotaxis* seeds were collected from mature plants in the field and 20 seeds were placed in each treatment (480 seeds) with an additional 82 seeds used in viability tests.

The viability of seeds was ascertained using 1% 2,3.5-triphenyltetrazolium chloride (Tetrazolium) and confirmed using 0.05% indigocarmine. A viable embryo stains red by Tetrazolium the false positives (dead embryos) stain with indigocarmine (Kearns & Inouye 1993).

9.2.3 Experimental before and after fire trials

As very little information exists regarding the fire responses and changes of the outcrop flora after fire, apart from anecdotal information, a before and after burn trial was necessary. With such a design data regarding the fire responses of individual species and the changes in composition, richness and diversity can be qualitatively collected and analysed. As fire responses of outcrop flora may be different from that of the

surrounding flora therefore the experimental design incorporates replicate trials off outcrops as well. The experimental design was limited to ground vegetation as larger fires would be unmanageable and trees are only a minor component of outcrop floras. To enable direct comparisons with the surrounding flora only ground vegetation (i.e. not trees) was burnt in these communities as well. As outcrop species are thought to respond to increased light and removal of competition (Chapter 1) one of the treatments in the experimental design included removal of above ground biomass. Thereby a test is provided that will differentiate between the changes noted due to fire and those due to increased light and removal of competition.

9.2.3.1 Location

Bald Rock National Park (BR) occurs in the extreme north east of the study region (Figure 2.1; Figure 9.1) and lies approximately 20 km north of Tenterfield in northern New South Wales. The park adjoins the New South Wales and Queensland State border. This region was chosen for fire experimentation, as it constituted an easily accessible and unmodified landscape with large and distinct bornhardts. The study area was also anthropologically unmodified and distant from public usage areas and therefore was secure against tampering. The bedrock is Stanthorpe Adamellite. The topography of the area is generally undulating to low hilly with intervening wide and flat swampy valleys (McDonald *et al.* 1995). The experimental fire plots were placed in close proximity to Carol's Creek Fire Trail and the Border Fire Trail (Figure 9.1)

Fires, both natural (e.g. lightning) and anthropogenic, are a common feature of Bald Rock and Girraween National Parks (Hunter 1999a. A number of high intensity fires have occurred within these parks in the past two decades. The largest of these recent fires occurred in December of 1994, with an estimated 90% of Girraween National Park being affected while a smaller proportion of Bald Rock National Park was burnt (Hunter 1999a).
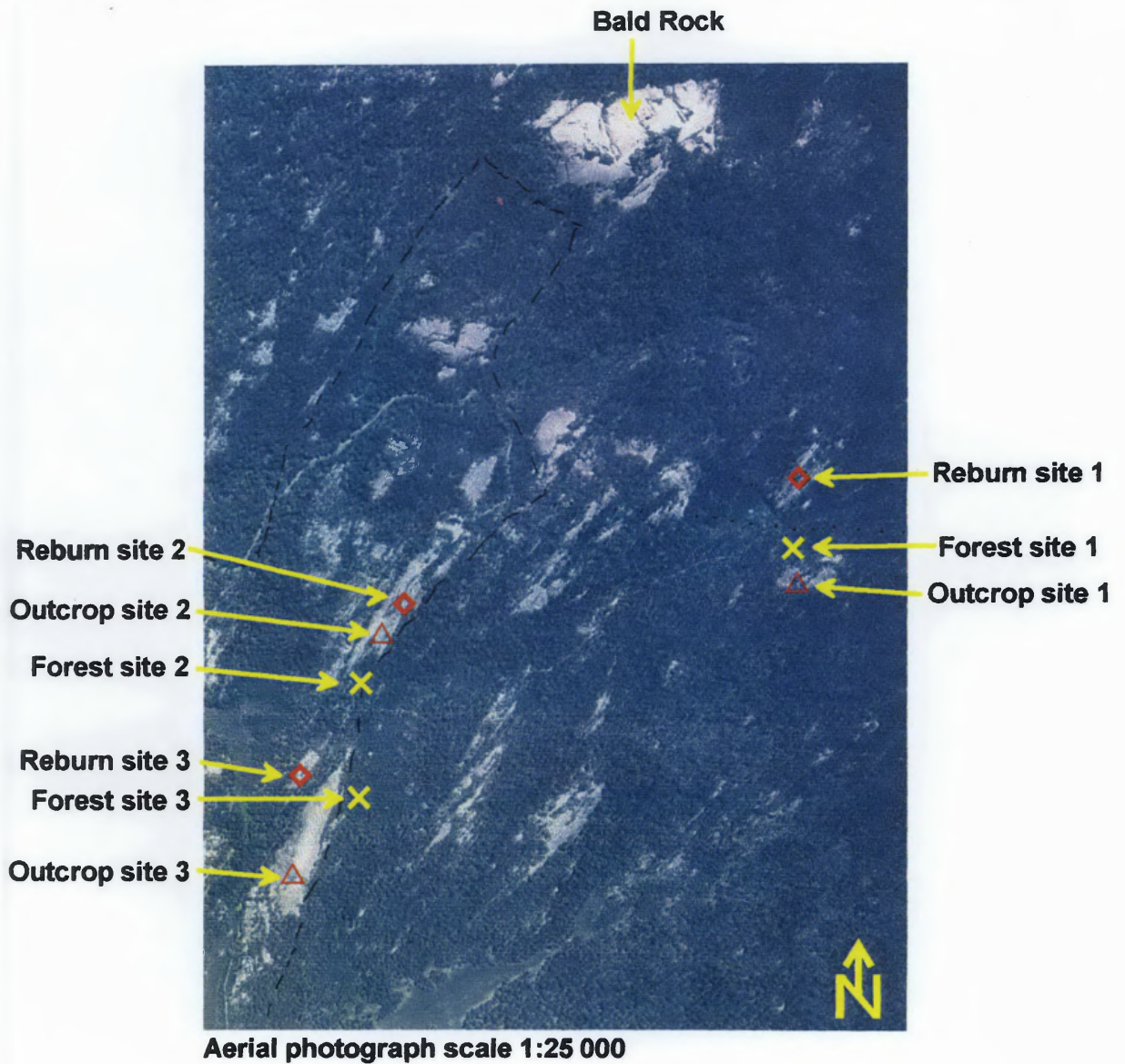
Aerial photograph scale 1:25 000

Figure 9.1: Location of fire experimental sites within Bald Rock National Park in Northern New South Wales. Black dashed line represents approximate New South Wales and Queensland State border.

9.2.3.2 Experimental design

Three separate fire experiments were conducted between December 1995 and January 1996. Habitats chosen for investigation included granitic outcrops that had no evidence of previous fires, outcrops that had been burnt one year previously (December 1994) and adjacent areas of forest (Figure 9.1 & 9.2). Each of the fire trials incorporated a before and after control incident design (BACI) (Underwood 1992) whereby data was collected before the applied treatment and afterwards once a month for 12 months.

Experiment 1

Three separate outcrops were chosen for investigation (Figure 9.1). On each outcrop, six 2 m by 2 m plots were randomly placed within a vegetated patch. Three experimental treatments; clearing, burning and control were duplicated within the six plots by random selection (Figure 9.2). All above ground biomass was removed from the cleared sites beyond the boundaries of the plot. Plots that were to be burnt had all above ground biomass alighted by drip torches to the point where no above ground biomass was observable. Control plots were left unmodified (Figure 9.2).

Experiment 2

Three separate outcrops were chosen (Figure 9.1) that had had their standing biomass burnt during the fires of December 1994. Within each of these outcrops, 2 m by 2m plots were randomly placed and two were allocated to each of two treatments, burning and control. The same procedures as Experiment 1 were used.

Experiment 3

Three forested areas were selected in close proximity to the Experiment 1 sites (Figure 9.1). Within each site, four 2 m by 2 m plots were randomly placed and two were allocated to each of two treatments, burning and control. The same procedures as given in Experiment 1 were used.
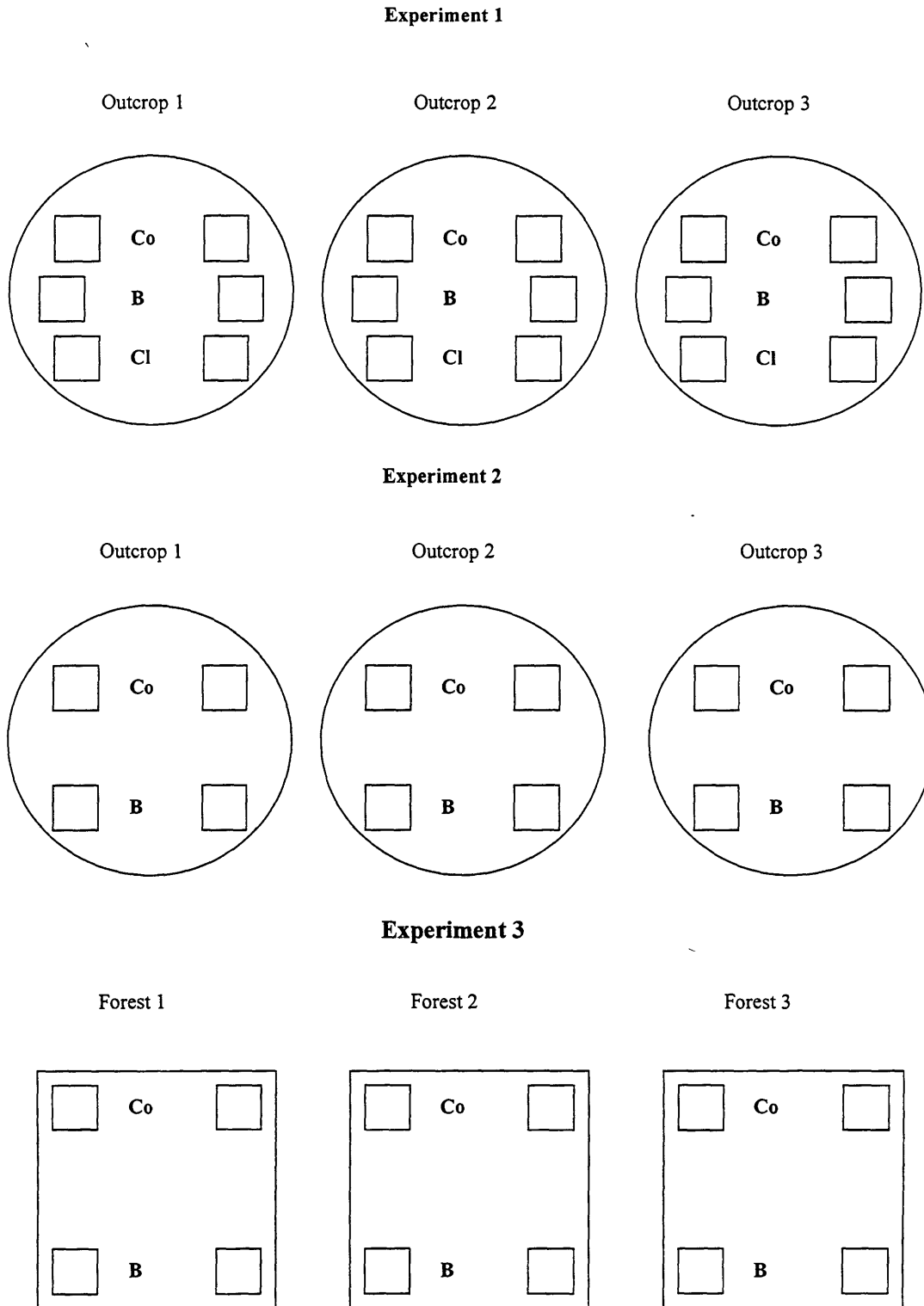
Experiment 1

Outcrop 1 Outcrop 2 Outcrop 3

Experiment 2

Outcrop 1 Outcrop 2 Outcrop 3

Experiment 3

Forest 1 Forest 2 Forest 3

Figure 9.2: Graphic representation of experimental design. Experiment 1 includes three treatments times two on three separate unburnt granitic outcrops. Experiment 2 includes two treatments times two on three separate previously burnt outcrops. Experiment 3 includes two treatments times two within three adjacent forest patches. Co = Control treatment; B = Burning treatment; Cl = Clearing treatment (Section 9.2).

9.2.3.3 Survey methodology and data collection

Each of the 42 experimental plots was 2 m by 2 m in area. The corners of each plot were marked by of 1.5 m tall wooden stakes. A 2 m by 2 m 'grid' of durable cord was created, consisting of 16 squares each measuring 50 cm by 50 cm. This grid was hooked over each of the four corner posts (Figure 9.3) and moved, up or down, depending on the height of the plants. Every plant species present within each plot was given two scores out of sixteen: a relative abundance score, based on presence or absence of primary rooting in each of the sixteen sub-squares, and a cover score, based on presence or absence of projective foliage cover. The size of each plot enabled the whole plot to be viewed equitably with little distortion, allowing accurate scores to be recorded for each species.
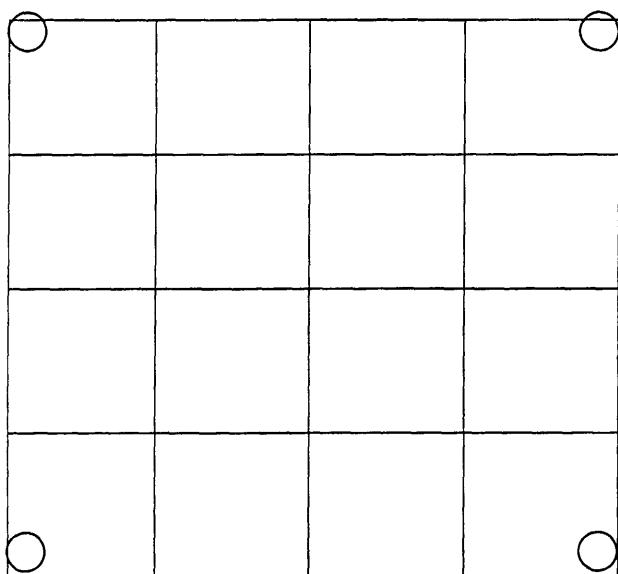

Figure 9.3: Diagram indicating Grid made to fit over corner posts of experimental plots enabling cover and frequency counts to be made for each species found.

The 42 experimental plots were surveyed once before treatments were applied and thereafter once a month, when possible, for a period of one year. During May of 1996, extensive rains and flooding occurred in the area making access impossible. In both September and November of 1996 extensive fires occurred in the general vicinity of the experiment making access to plots dangerous. All data were entered on a relational database in Paradox 7 for Windows as discussed in Section 2.5.

9.2.3.4 Multivariate and univariate analyses

Richness and diversity were calculated for each plot (42 actual plots) at each monthly recording session (402 temporal plots in total). Richness is the number of species in each plot and diversity was measured by the Simpson diversity measure. Diversity measures differ primarily in the degree to which they emphasize species richness versus species evenness. In temporal comparisons the number of species is often of less importance, compared to, relative abundance, due to competitive displacement or extinction. When changes in relative abundance is of importance Simpson's D is an appropriate measure of diversity as it emphasises evenness (Huston 1994). Krebs (1985) considered that such a method summarises most of the biological information on diversity. The relative cover score out of 16 was used in all calculations as a direct substitute for abundance. Richness and diversity scores over time were plotted using Excel 7 for Windows. Before and after (Px [initial] − Po [final]) richness and diversity scores were used in ANOVA analyses conducted using STATISTICA (StatSoft 1994-1995). Species accumulation (richness) over time was also calculated for each treatment. Species accumulation was along with richness per unit time was used to give an indication of species turnover over the period of the trial.

The PATN Analysis Package (Belbin 1995a, b) was used for classification and ordination. All species and their relative cover scores were used in the classification analyses. The Kulczynski association measure, which is recommended for ecological data (Belbin 1995a, b), along with flexible UPGMA and Semi-Strong-Hybrid multidimensional scaling (SSH) were used. The default PATN settings within these options were used. For a further discussion of these methods see Section 3.2.2.

'Canonical Correspondence Analysis' (CCA) via CANOCO (ter Braak 1987-1992), a direct gradient analysis technique, was used for exploration of factors affecting the composition of plots. Environmental variables for analysis included treatments and site location. Forward selection of variables was used for data reduction, ranking of variable importance and significance testing (ter Braak & Verdonschot 1995). Monte Carlo tests were performed on the results. For further discussion of these techniques see Section 3.2.2.

9.3 Results

9.3.1 Autecological observations

Dominant species found on granitic outcrop sites where observations were made prior to the 1994 fires, were no longer present on burned sites after the fires (Figures 9.4 & 9.5). A number of taxa considered rare nationally (some of which are endangered or vulnerable in New South Wales), or uncommon, was found to be prolific and dominant after the passage of fires on outcrops. These species were not found in any of the numerous adjacent unburned granite outcrops or unburned vegetation patches on the same outcrops. Notable examples of species found to be dominant on outcrops after fire along the eastern portion of the batholith include; *Acacia adunca* (Figure 9.4), *A. latisepala* (Figure 9.5 & 9.6), *Actinotus gibbonsii*, *Bulbostylis densa*, *Micrantheum hexandrum*, *Monotaxis macrophylla*, *Muehlenbeckia costata*, *M. rhyticarya*, and *Pelargonium australe*. The species of *Acacia* are relatively short lived (10—20 years) compared to most perennial outcrop restricted taxa (50 years +) (*pers. obs.*). Many of the other species are short-lived ephemeral species (*pers. obs.*). Despite these observations a number of outcrops and patches of vegetation on the same outcrops often showed no development of any of these taxa. No large fires occurred in the western portion of the batholith, however, within small lightning ignited fires on outcrops, similar observations were made. Intensive observations were made on *Muehlenbeckia costata* and *Monotaxis macrophylla*.

Muehlenbeckia costata populations were only found on granitic outcrops burnt in the 1994 fires at altitudes above 1100 m. Populations occurred within Bald Rock and Girraween National Parks (BR) and Butterleaf (BL) and Warra State Forests (BC). Prior to these fires no individuals had ever been recorded in Bald Rock National Park or Butterleaf State Forest. Adjacent unburnt patches on the same outcrops had no germination of this species. In addition, this species was only found on the largest of the rocky outcrops within any region, and these outcrops were mostly over 10 ha in area. When *M. costata* was found on smaller outcrops, these were satellite outcrops in close proximity to the larger outcrops bearing populations. In no instance has a single germination ever been seen in localities off outcrops.

In early February 1995, only four months after the passage of wildfire, individuals of *M. costata* had already spread 2 m in all directions from the central root and were flowering. Flowering was noted in all individuals and fruiting in all female plants on all occasions, including during all monthly visits to populations within Bald Rock National Park, and it is assumed that flowering and fruiting are continuous from about three months after germination until senescence. Individuals of this species were found to extend up to 5 m from the central rootstock in any direction and clamber up the stems of nearby plants to a height of 4 m. A single plant is able to dominate an area 10 m in diameter and 4 m vertically when mature (Figure 9.6).

By June of 1996, most plants in all localities had senesced, almost two years after original germination. Individuals that had not died were present only as small *c.* 20 cm long shoots, resprouting at the central taproot. Persistent plants were still flowering, but a heavy infestation of a rust fungus was noted on all plants immediately prior to, and during, the senescence of plants.

Large populations of *Monotaxis macrophylla* were found within Warra State Forest (Backwater: BC) in February 1995. This species was restricted there solely to burnt patches of vegetation (1994 fires) on granitic outcrops. No plants were found within unburned vegetation on granitic outcrops or in the surrounding vegetation. *Monotaxis macrophylla* had not previously been found on the Northern Tablelands of New South Wales and was not obviously present on the same outcrops in Warra State Forest on visits only six months prior to this visit. By the end of 1995, no trace was left of the populations and none was found during two years of periodic observations in the area in the absence of fire.

Figure 9.4: *Kunzea bracteolata – Leptospermum novae-angliae* heath prior to fire has been replaced by *Acacia adunca* after the passage of fire. Girraween National Park (BR).

Figure 9.5: *Leptospermum novae-angliae* dominated heath prior to fire replaced by *Acacia latisepala* (14 months after original fire). Bald Rock National Park (BR). The bare patch in the centre is where one of the reburned 2 m x 2 m plots was placed.

Figure 9.6: A burnt crevice on a large outcrop at Bald Rock National Park (BR) 12 months after fire. This crevice was dominated by *Leptospermum polygalifolium* subsp. *montanum* prior to the fire and is now dominated by *Muehlenbeckia costata, Acacia latisepala* and *Helichrysum boormanii*.

9.3.2 Seed germination

Viability tests indicated that 63% of *Muehlenbeckia costata* and 88% of *Monotaxis macrophylla* seeds were viable. Despite the experimental treatments, no germination was recorded for *Monotaxis macrophylla* seed. A total of 34 seeds (6%) of *M. costata* germinated. Due to the low number of germinations the results of treatments were pooled into two treatments 'smoked' and 'non-smoked'. Twenty-nine seeds germinated after being treated with smoke (17 after scarification and heating to 80°C) with only five seeds germinating without smoke treatment all of which were also unscarified and not heat treated. However, the null hypothesis that germinations were not different could not be rejected as the differences were found to be insignificant ($P = 0.072$; t test). Plants growing from those seeds that germinated flowered within 3 months after radicle appearance (plants 5 cm in length) and continued to flower until senescence at around 18 months.

9.3.2 Multivariate analyses on species composition

The floristic species composition of outcrops before and after treatment and on previously unburned, cleared and reburned plots is significantly different from that of any of the forest plots (Figures 9.7 & 9.8). The first division on the dendrogram is between forest plots and outcrop plots (Figure 9.7). This is then followed by a less distinct separation between the two experiments conducted on outcrops. Temporal plots from each site and treatment clustered together within the analyses.

The ordination analysis and scatter plot (Figure 9.8) clearly show the forest sites distinctly allied to each other and separated from the outcrop sites. Although there is evidence to suggest that the two experiments conducted on outcrops were separated in the ordination, this is less distinct. Some widely separated sites (outliers) within the scatter plot are evident and all are from the reburned fire trials on outcrops indicating that this experiment led to some major changes in floristic composition and abundance over the period of the trial. The results of the ordination corroborate those of the dendrogram indicating that the results are robust.

Overall, Monte Carlo significance testing (999 iterations) of the first canonical axis and the overall test on trace are highly significant ($P<0.001$) for all three CCA's. The first canonical axis accounts for 77.9%, 64.7% and 55.6% for Experiment 1, Experiment 2 and Experiment 3 respectively. Site locality was the strongest explanatory component of species composition and cover, along with two treatments of more minor explanatory power in each of the CCA's conducted on the three experiments (Figure 9.9). The correlation of site locality and composition is more clearly discernible in Experiment 2 than Experiment 1 and more effective in outcrop sites in general than forest sites (i.e. as indicated by the clumping of sites along the locality vectors) (Figure 9.9).

The treatments of 'control' and 'burning' were significantly correlated with the CCA ordination of Experiment 1. The treatments of 'clearing' and time-since-fire (TimeFire) were insignificant in this analysis. The 'burning' treatment was more effective within plots on Outcrop 2. 'Control' plots were more distinct from cleared or burnt plots within Outcrop 3.

In addition to site locality, the 'control' treatment and the recorded time-since-fire (TimeFire) were significant in explaining ordination position. However the 'burning' treatment was insignificant. Both 'control' and time-since-fire (TimeFire) were more positively correlated with Site 3.

The CCA ordination of the forest plots indicates that the treatment of 'burning' and, less significantly, time-since-fire (TimeFire) were important in site distribution. Unlike the outcrop plots however, the 'control' treatment was insignificant in analyses of forest plots.
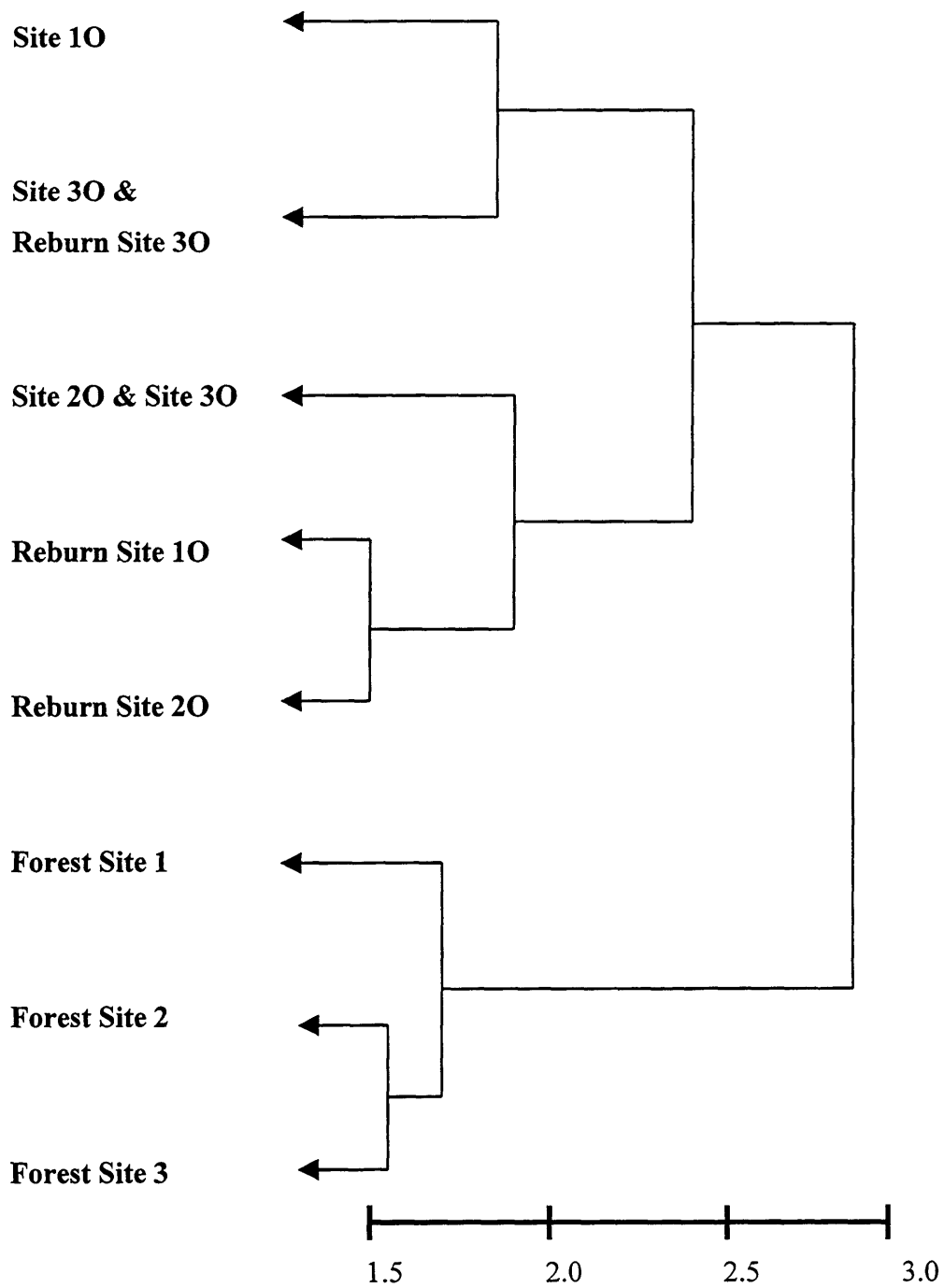
Site 1O

Site 3O &
Reburn Site 3O

Site 2O & Site 3O

Reburn Site 1O

Reburn Site 2O

Forest Site 1

Forest Site 2

Forest Site 3

1.5   2.0   2.5   3.0

Figure 9.7: Summary dendrogram of eight floristic groups based on Kulczynski association and flexible UPGMA fusion classification of all temporal fire experimental plots (402 in total) (Section 9.2.3.4). The full dendrogram is given in Appendix G.
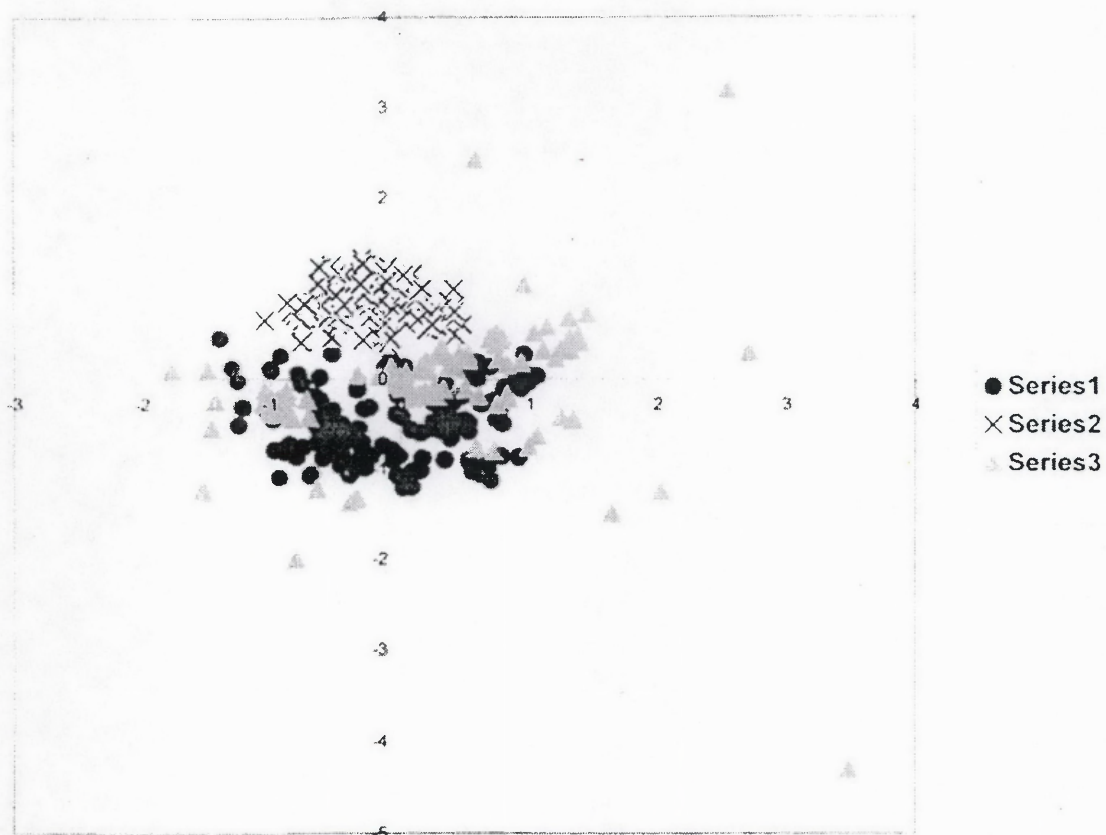
Figure 9.8: Ordination scatter plot of all temporal plots (402) based on full floristics and analysis by Flexible UPGMA association measure and Semi-Strong-Hybrid Multi-dimensional Scaling (Section 9.2.3.4). Series 1 = burned outcrop plots; series 2 = burned forest plots; series 3 = reburned outcrop plots.
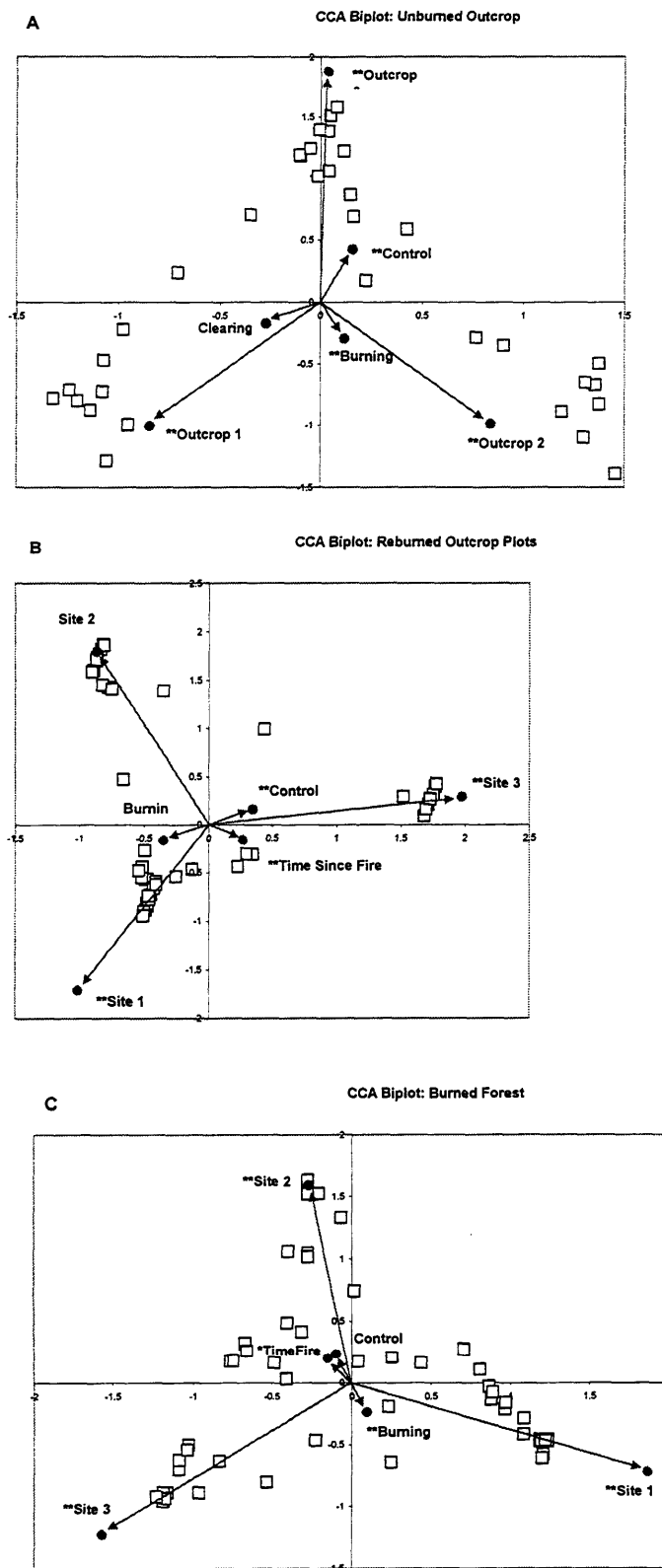
Figure 9.9: Biplots of CCA results of the significant variables chosen by forward selection and Monte Carlo significance testing (Section 9.2.3.4) against sites (squares). Significance of variables based on Monte Carlo simulations are; **P<0.001, *P<0.01, no * variable insignificant. A) unburned outcrop plots. B) reburned outcrop plots. C) burned forest plots. The length of the lines indicates the strength of the relationships.

9.3.3 Univariate analyses on richness and diversity

ANOVA analyses performed on species richness and diversity before and at the end of the experimental trials (Po – Px) were insignificant and are therefore not presented. Overall changes in species richness and diversity in each of the experimental trials are given in Figures 9.10 and 9.13.

Species richness declines across most replicates in all experiments initially after burning (Figure 9.10a, 9.12a, 9.13a). The initial drop in species richness is most marked in Experiments 1 and 2 (outcrop plots) and in particular in Experiment 2 (outcrops previously burned the year before). The difference between the initial and end species richness in burning treatments is mixed and non-significant. However, in most treatments richness returns to a level comparable to pre-treatments. Species diversity as measured by Simpson's *D* in most instances does not return to the pre-burning treatment diversity in Experiments 1 and 2 (outcrop plots) but appears to be more variable in Experiment 3 (forest plots). Species accumulation occurs in most plots over time after burning. This is most pronounced in Experiment 3 (forest plots) (Figure 9.13).

Clearing as a treatment only occurred in Experiment 1. Although non-significant, most plots decreased in richness by a small amount over the period of the trial. A sudden initial decrease in richness after treatment was not as apparent as in the burning trial. Although species richness overall decreased in most plots, some species accumulation still occurred over time. Species diversity changed throughout the period of the experiment in this treatment.

The species richness of control treatments changed over time in most plots. This was less pronounced in Experiment 1 but more so in Experiment 2 and clearly so in Experiment 3. In Experiment 1, species diversity is highly variable over time, in contrast to species richness. Variability in species diversity although less pronounced also occurs in Experiments 2 and 3. Species accumulation increased in most control treatment plots over time. However this was less pronounced in Experiment 1, where at least half of the plots had no increase in species numbers.
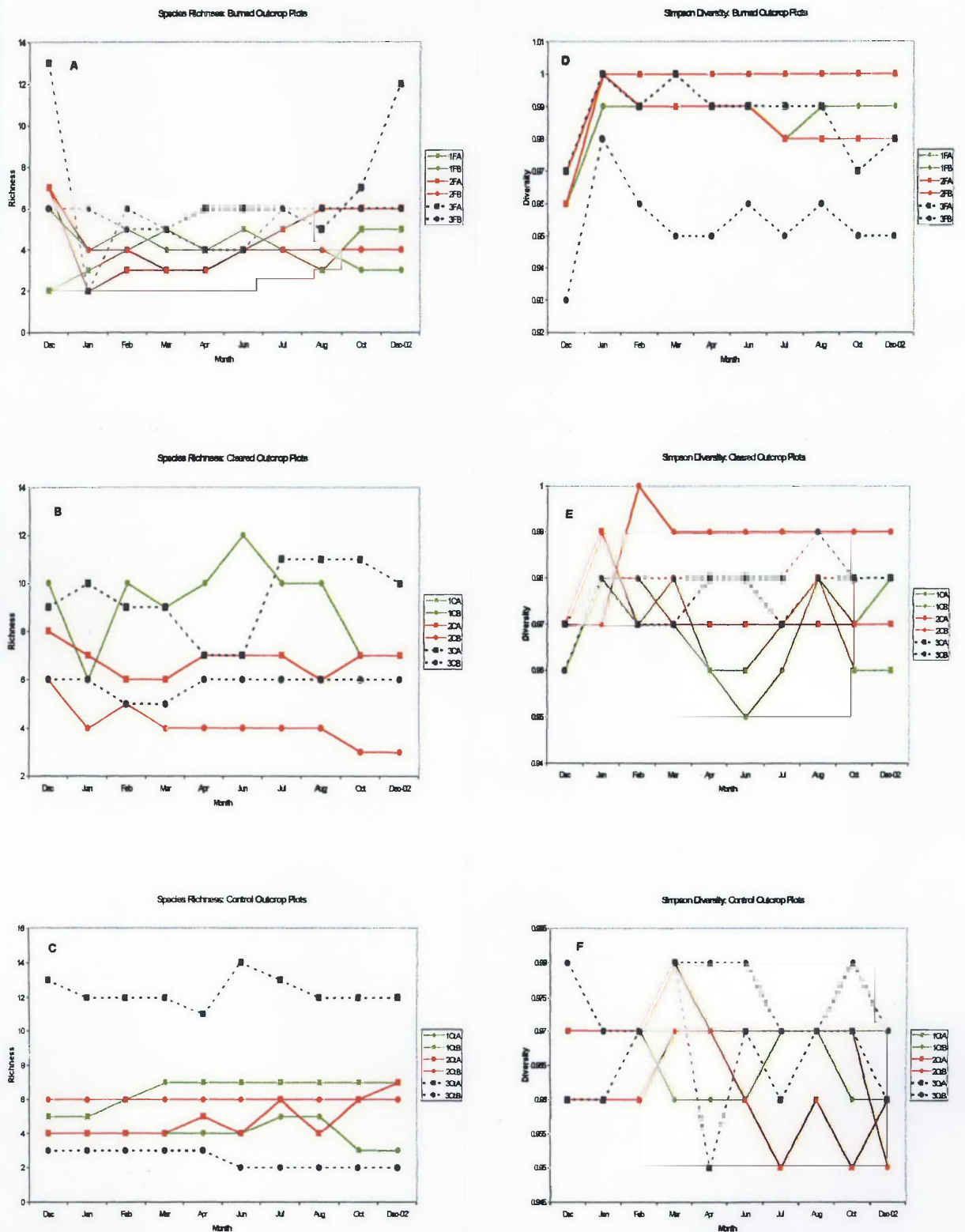
Figure 9.10: A-C; species richness over time in each of the three fire trial treatments on outcrops (Experiment 1). D-F; species diversity as measured by the Simpson's *D* Index over time in each of the three fire trial treatments on outcrops.
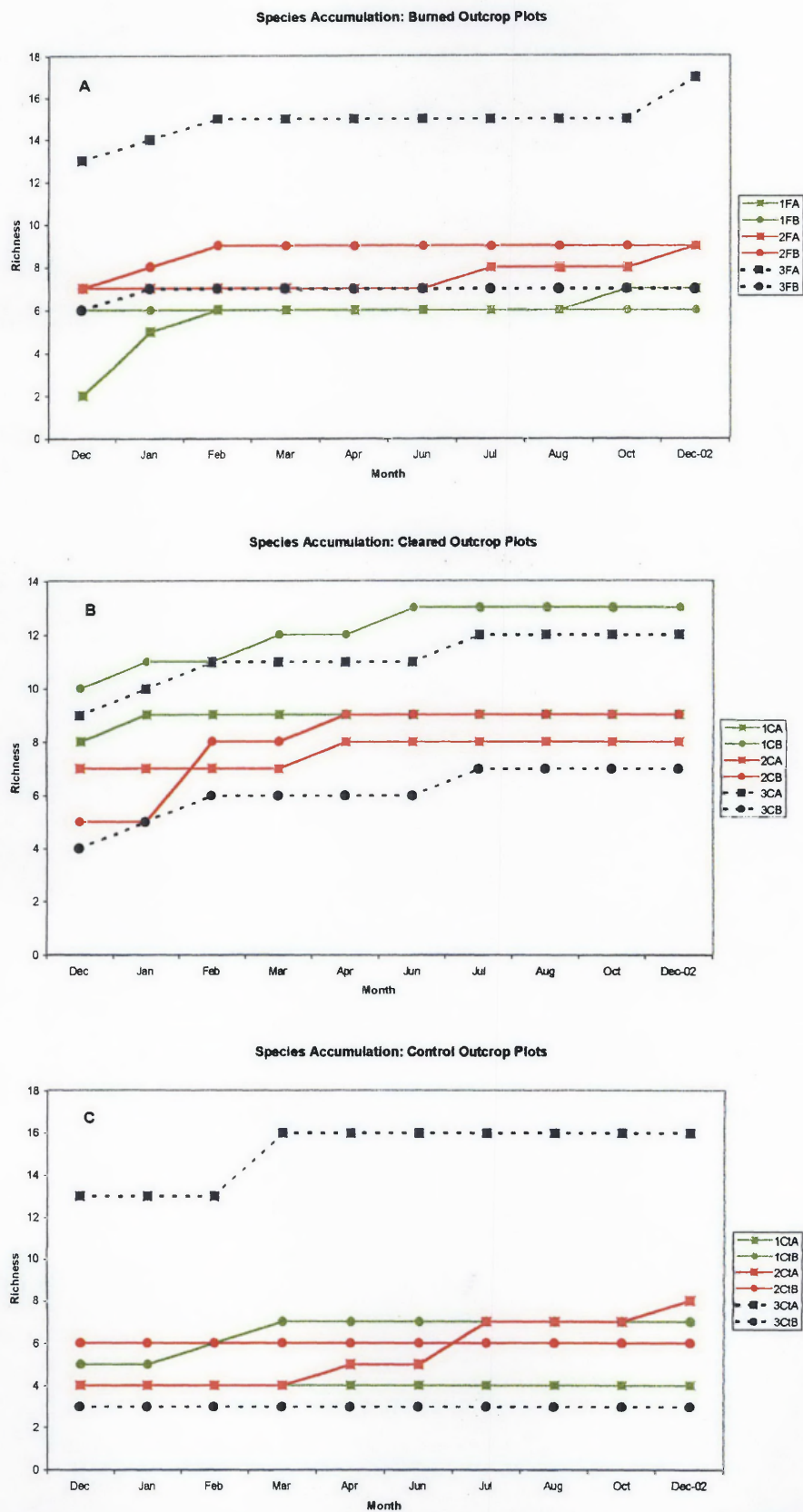
Figure 9.11: A-C, cumulative number of species found in each of the three fire trial treatments on outcrops (Experiment 1).
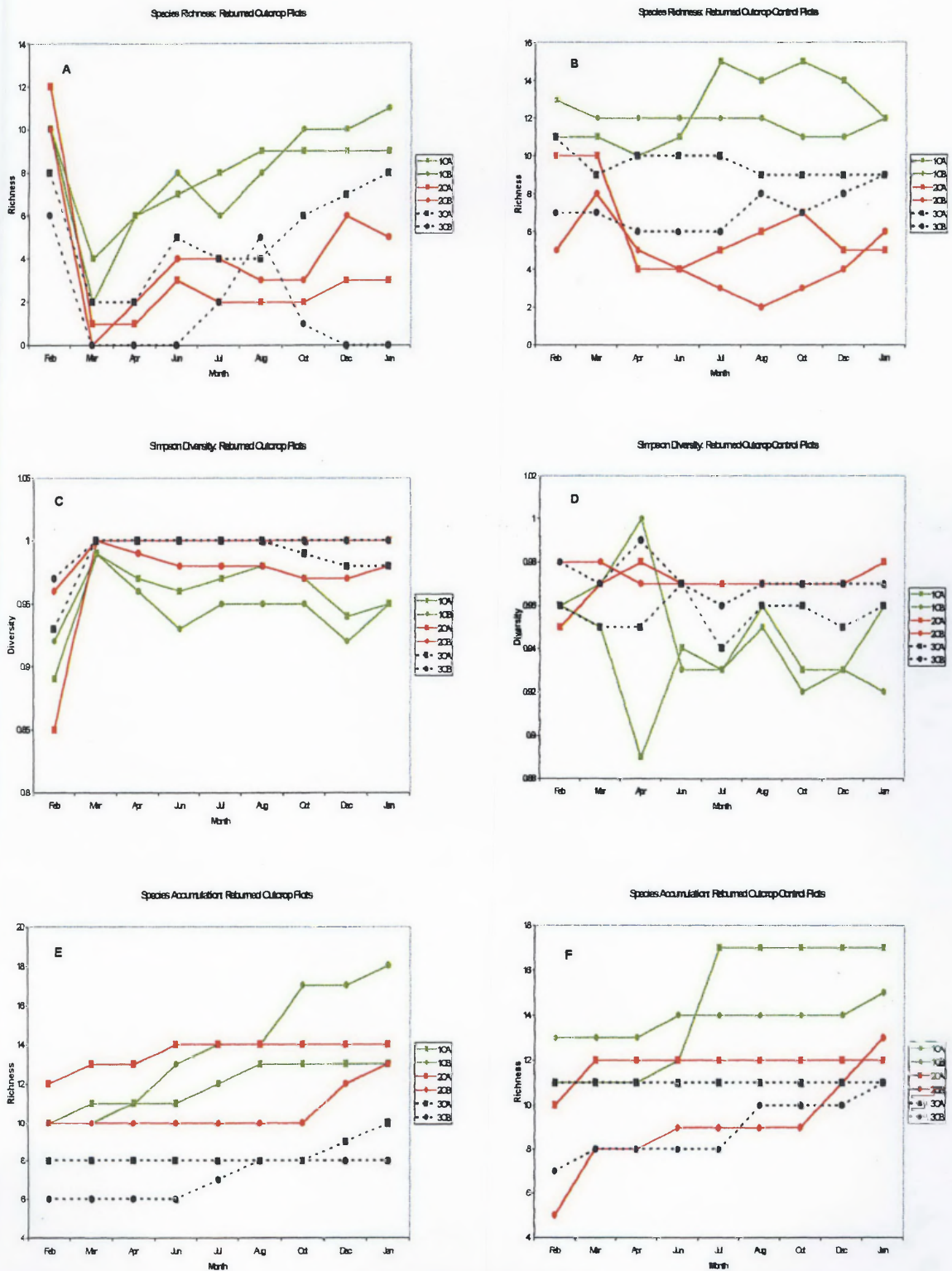
Figure 9.12: A-B; species richness over time within the two treatments on reburnt outcrop. C-D; Simpson's *D* over time within the two treatments on reburnt outcrops. E-F; cumulative number of species over time within the two treatments on reburnt outcrops (Experiment 2).
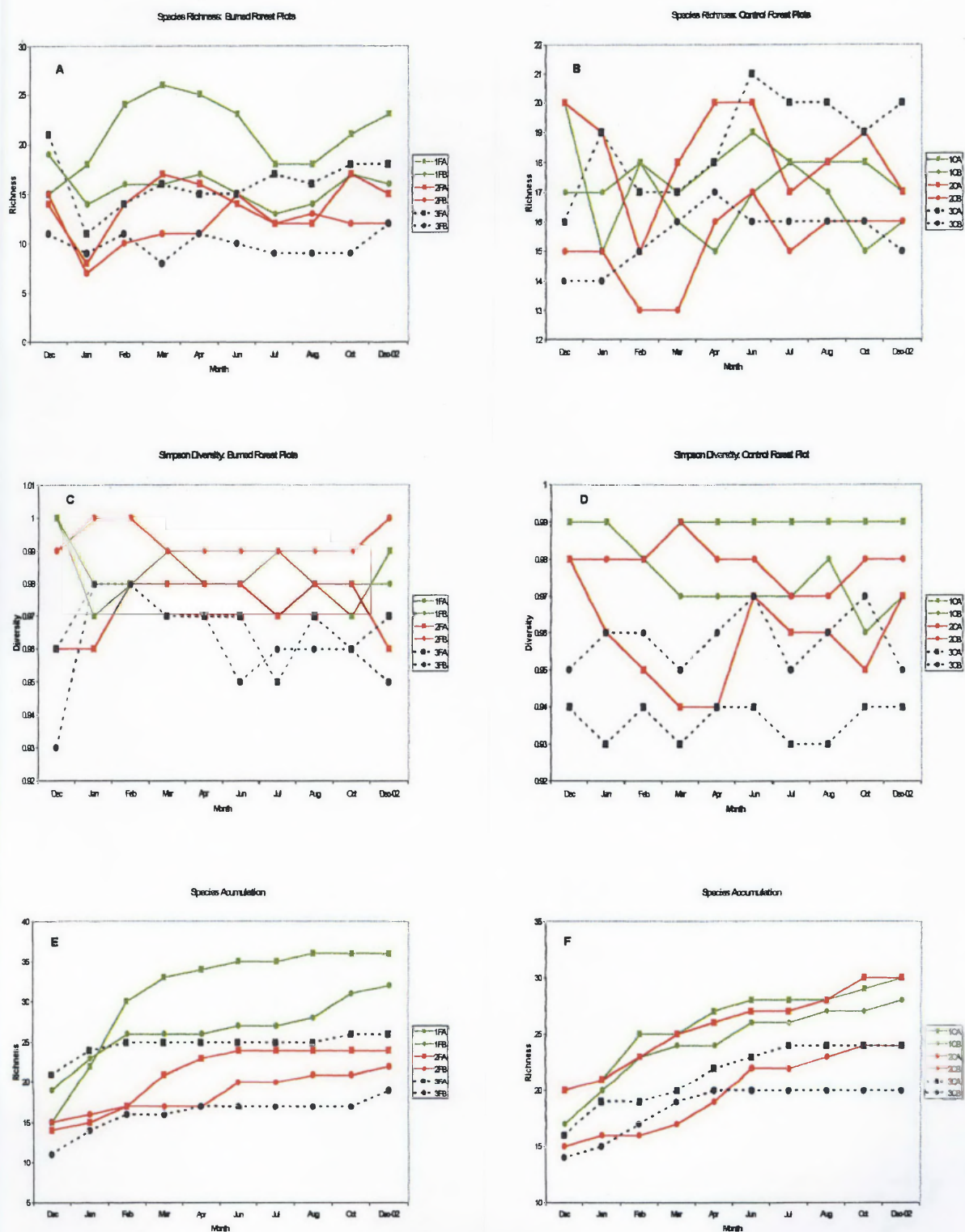
Figure 9.13: A-B; species richness over time within each of the two treatments within forest plots. C-D; Simpson's *D* over time within each of the two treatments within forest plots. E-F; cumulative species richness over time within each of the two treatments within forest plots (Experiment 3).

9.3.4 Regeneration strategies between habitats

Species have two major responses to fires, namely obligate resprouting or obligate seeding (Gill & Bradstock 1992). Resprouting species are believed to be more commonly associated with fire prone environments and obligate seeders with environments with a low fire frequency (Gill & Bradstock 1992). The responses of species to fire in terms of these two responses were collected via anecdotal observations (Section 9.2.1) and during the fire trials (Section 9.2.3). This information is held in an unpublished interactive identification database (DELTA) (Dallwitz 1980; Dallwitz *et al.* 1993) of granitic occurring species created by the author. The differences in the percentage of obligate seeders on outcrops and in the surrounding forests sampled at Bald Rock National Park (BR) are presented in Figure 9.14. There is a substantially greater percentage of obligate seeders in the flora of granitic outcrops as compared to the flora that surrounds outcrops.
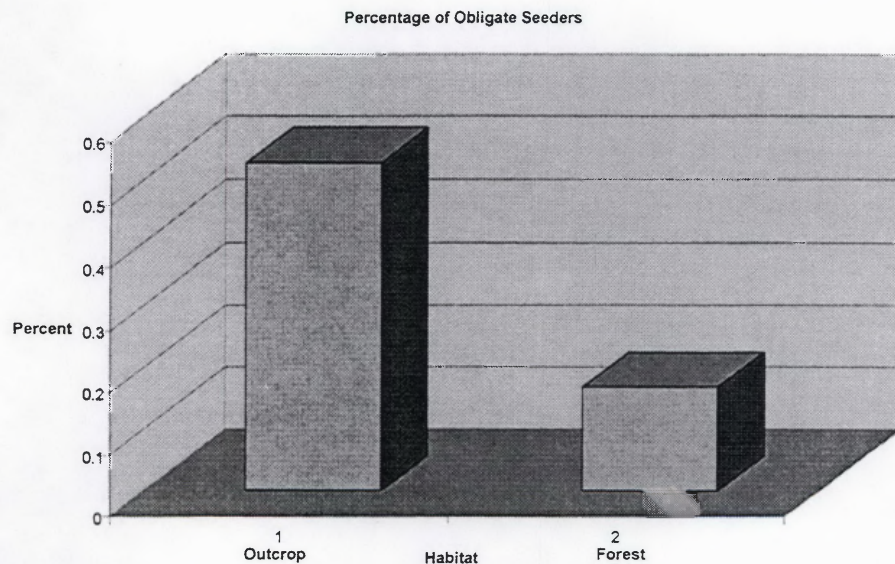

Figure 9.14: Overall percentage of obligate seeders represented in the outcrop and forest floras sampled during the fire trials at Bald Rock National Park.

9.4 Discussion

9.4.1 Implications of multivariate and univariate results

The results of Chapter 3 and 6 have confirmed that the flora of granitic outcrops is distinct from that of the surrounding flora. Although this distinctiveness varies in degree across the batholith, it is at its highest in the north east at Bald Rock and Girraween National Parks (BR) and along the eastern escarpment in general. This difference is mirrored in the results of the experimental plots. Both cluster analysis and ordination of the 402 temporal fire plots show the flora within outcrop plots is distinct from those of the surrounding forests. This difference is found even after burning and clearing treatments. The distinctiveness of the flora in Experiment 3 from forest plots was evident even with fires occurring in two consecutive years (one natural and one experimental). However, published evidence suggests that under a continuous frequent burning regime the flora of outcrops may lose their distinctiveness. For example, Binns (1992) found that a high frequency of fire on granitic outcrops, in the same region, caused heaths that were dominated by outcrop endemic taxa to be reduced to grasslands and herbfields dominated by ubiquitous species such as *Lomandra longifolia* and *Imperata cylindrica*. Hunter *et al.* (1999) has made similar anecdotal observations at Demon (DM).

Of particular note in the dendrogram and scatter plot (Figure 9.7 & 9.8), is the groupings of temporal plots in analyses, despite the treatments imposed. This result is complemented in the CCA biplots (Figure 9.9) which show site locality as the most significant and strongest explanatory variable. It can be concluded that, regardless of treatment, initial floristic composition is the most important factor in structuring the composition after the imposition of each treatment. In addition, as site was the most important factor in each analysis, and plots from each site largely grouped together, the surrounding species pool is also of considerable importance. Responses to the treatments imposed were individualistic and based on the surrounding species pool available at each site (outcrop or forest area) and the initial composition of each plot. Similar results have been obtained in many Australian systems and illustrate the 'initial

floristic composition' model (Engler 1954), where the initial species composition after a
disturbance determines the subsequent composition (Purdie 1977; Noble & Slatyer
1981; Clark 1988; Williams & Gill 1995). This is a surprising result as all sites are
within relatively close proximity, being within only a few kilometres of each other and
subsequently within the same Element and community (Chapter 4; Community 2d).

The individual treatments (burning, clearing, control and time-since-fire) varied in the
strength and significance of their effects at different localities. This indicates an
individualistic response to the treatments at each site that is due to initial species
composition, thus reinforcing the above observations. Formulating strategies or
responses of communities with unsurveyed floristic composition are almost impossible,
regardless of their overall affinities or structure. On the other hand if initial species
composition is known and sufficient information exists on species responses to
treatments (such information does not exist at present), reliable predictions may occur.

In Experiment 1 treatment of clearing had no significant effect, with control and burning
being significant. This may be due to mixed responses of species to being severely
pruned with some dying and others recovering by resprouting (Figure 9.10b).
Comparatively little overall change occurred in the composition of the control plots
making them significantly different from the other treatments (Figure 9.10c & 9.11c).
Time-since-fire (TimeFire) was insignificant in Experiment 1 probably due to the
overriding effects of the treatment burn. Control and time-since-fire were significant
treatments in Experiment 2 with the burning treatment being insignificant. This is
probably due to sites being burned only one year previously and therefore, reburning
sites had little effect except in reinforcing the overall changes due to the initial fire. The
scatter of plots in the CCA biplot of Experiment 3 (forest plots) is greater and less
clumped than in the Experiment 1 or 2, indicating that although initial site composition
is of overriding importance, it is comparatively less important than in the outcrop plots
(Experiment 1 & 2) (Figure 9.9). The scatter of plots may have been affected by the
overall greater effect of fire on forest plots as indicated by both time-since-fire
(TimeFire) and burning both being significant in analyses.

No significant results were obtained among analyses of the difference between the
initial and subsequent species richness and diversity. Therefore the null hypothesis of no

difference in germination responses could not be rejected. The differences tested were insignificant due to the highly individualistic and stochastic responses of the plots to each treatment. Although control plots were less stochastic overall, changes occurred with no consistency between replicate plots in their responses. Within Experiment 1 species accumulation was low over time indicating little species replacement after treatment and over time. The control plots in Experiment 1 had at least half of the plots with no species replacement at all (Figure 9.11c) and yet the species diversity changed in all, indicating little change in composition and also richness, but many changes in dominance (Figure 9.10f). Grubb (1986) found this to be common place in communities and termed such changes in abundances 'drifting clouds of abundance'. Conversely, the control plots of both Experiment 2 and 3 showed changes both in species richness over time and in species replacement in almost all plots, with dominance changing in all (measured by species diversity).

It is likely that the small size of the experiment reduced the likelihood of a significant result (Bellehumeur *et al.* 1997). Regardless, it is of note that no consistency was found in any of the treatments over time, with even control plots showing changes in species number, abundance and composition over time, albeit inconsistently. Such results reinforce those found for multivariate analyses in indicating the highly individualistic changes and responses of plots over time. This is likely to be due to the highly stochastic composition and dominance of species at individual sites. Even though there was no consistent increase or decrease in richness and diversity over time between or within treatments, species richness and diversity was maintained consistently close to the pre-treatment level. This indicates that some inherent structure exists at each site for a predetermined level of richness and diversity. This may once again be due to the interaction of species present that are largely the same before and after treatments (initial floristic composition model).

It is apparent that there is a great inherent variability in responses that are based on initial composition and individual site characteristics. Such inherent variability has been found consistently in studies of granitic outcrops and their component floras at all levels (Section 1.5; Section 4.4.4). Such responses would enable the maintenance of a high level of biodiversity and richness on a habitat (beta diversity) and landscape (gamma

diversity) scale. Richness and diversity however, would be limited on the local scale (alpha diversity) by initial composition and the available species pool.

The results presented here are in marked contrast to a number of anecdotal observations made by other researchers (Binns 1992; Hunter *et al.* 1999) and those presented here that indicate marked changes in composition. Such inconsistencies may not be contradictory. The small scale and size of experimental plots, their placement and the intensity and temperature of the fires may easily account for such differences. Bradstock and Auld (1995) have shown that low-intensity fires may be detrimental, as the heat may be insufficient to stimulate the germination of buried and dormant seeds. Even after the very large 1994 fires discussed here, a number of vegetation patches did not change dramatically in their composition even when they occurred on the same outcrop as patches that did.

9.4.2 Fire promoted flora and granitic outcrops

Autecological observations, both qualitative and experimental, provide sufficient evidence that a fire-ephemeral flora exists on granitic outcrops within the New England Batholith, at least in some localities. Similar findings have occurred within the outcrop flora of Western Australia where it is believed that up to 30% of species only appear after fire (Stephen Hopper, *pers. comm.*, Nov 1998). Fire-ephemeral species have been noted in many communities and occur across a range of plant families (Gill 1993). The appearance of herbaceous fire-ephemeral species has been noted around the world (Thanos & Rundel 1995) and in other heath and shrubland communities within Australia (Gill & Groves 1981; Gill 1993). Certain lifeform traits are thought to be associated with fire-ephemeral taxa which have evolved in fire prone environments: I) germination stimulated by the passage of fire, II) individual plants have a short life span, III) a large biomass produced in a short period of time, IV) flowering occurring shortly after germination, and V) non-persistent populations in the absence of fire. Many of the taxa found on granitic outcrops after fire possess such characteristics (Section 9.3.1).

Muehlenbeckia costata may be considered a fire-ephemeral species. A single plant may dominate an area of 10 m in diameter and 5 m in height in as little as 1 year. The population size as given by Hunter *et al.* (1998) changed from zero to 1200 to zero individuals in a matter of three years within Bald Rock National Park. A large number of seeds are produced over the life span of *M. costata*. A single plant may yield many hundreds of nuts at any one time, and production is continuous throughout the one to three year life of the plant. Much of the seed is viable. If such a large seed 'rain' is normal for this species, a limiting factor of population size must be the local dispersal distance. If dispersal was efficient, a larger number of burnt outcrops, including many of the smaller outcrops, should have had populations of this species and populations would be able to re-invade areas where it has become locally extinct. This was not the case. Such localised dispersal is typical of species in poorly connected systems (Gaston & Lawton 1990; Green 1994; Dieckmann *et al.* 1999). *Muehlenbeckia costata* has adaptations that allow the diaspore to be dispersed by vertebrates (fleshy, sweet, coloured expanded calyx), but which also allow it to withstand high temperatures associated with fires (thick, hard-walled nut). Germination was observed to occur after heat treatment of 120°C for ten minutes providing evidence that the nut can survived high temperatures associated with fires.

Monotaxis macrophylla responds in a very similar way and could also be considered a fire-ephemeral taxon. Within New South Wales this species has only ever been found on rock outcrops that have been recently burned (Hunter 1998b). This species was also found to increase in biomass enormously after fire, seed prolifically very shortly afterwards, and senesce in less that a year not to be found again. Other species have also been found to respond in a similar fashion for example *Actinotus gibbonsii* and *Muehlenbeckia rhyticarya* to name two (Section 9.3.1).

It may be expected that fire promoted taxa would occur in areas with a frequent occurrence of fire. However, granitic outcrops, especially if large, have been considered refuge areas for taxa that are not fire adapted (Gillham 1961; Ashton & Webb 1977; Craven & Jones 1991; Erickson *et al.* 1991; Fuls *et al.* 1992; Binns 1995a; Gröger & Barthlott 1996; Beard 1997; Hopper *et al.* 1997; Heinze *et al.* 1998; Hunter 1998a; Lawler *et al.* 1998; Hopper 1999). Fire occurs much less frequently on granitic outcrops

than in the surrounding areas. Even in the 1994 fires which burnt 90% of Girraween National Park (BR) and a similar portion of Warra State Forest (BC), up to half of the outcrop vegetation patches were unaffected, including many small outcrops of about 1 ha in size *(pers. obs.)*. Many of the vegetation patches on the New England Batholith are of a substantial age, and humus development is considerable. Some of the larger outcrops are refugial areas for rainforest taxa such as *Quintinia sieberi, Rapanea* spp., *Notelaea* spp., *Tasmannia glaucophylla* and *Trochocarpa laurina* (Chapter 3). Ashton and Webb (1977), working in south-eastern Australia, considered the intervals between fires on granitic outcrops, within a matrix of fire prone vegetation, would still be in the order of several centuries. Erickson *et al.* (1973) considered that inter-fire intervals on outcrops in the fire prone Western Australian south west were also very large as evidenced by the very thick trunks and relative size of shrubs. The floras of outcrops have a higher proportion of obligate seeders than many other temperate Australian communities (Gillham 1961; Erickson *et al.* 1973; Ashton & Webb 1977; Craven & Jones 1991; Fuls *et al.* 1992; Binns 1995a; Gröger & Barthlott 1996; Beard 1997; Hopper *et al.* 1997; Heinze *et al.* 1998; Hunter 1998a; Lawler *et al.* 1998; Hopper 1999) implying a low fire frequency (Figure 9.14).

Such observations suggest that fires are indeed much less frequent on outcrops and that even within fire prone environments they are likely to have fire intervals many times greater than the surrounding forested or woodland vegetation. They are thus unlikely to evolve a fire promoted flora. Only the most intense fires are likely to affect areas on the larger outcrops where many of the fire promoted taxa occur. It is therefore paradoxical that fire promoted taxa are not only restricted to granitic outcrops but are in many cases restricted to only the largest outcrops.

Species are not independent entities and commonly share adaptations through a common ancestry. It is, therefore, important to examine the phylogenetic distribution of traits (Harvey & Pagel 1991). Brandbyge (1992) for instance states that species of *Muehlenbeckia* are 'weedy', rapidly build up a large biomass, and are characteristically found in open, rocky, sun exposed habitats. Mallinson *et al.* (1998) has also found that often post disturbance recruitment with a long dormancy of propagules in the soil seed bank are common in *Muehlenbeckia* from various habitats. It appears therefore, that the

ecology of many species of *Muehlenbeckia*, apart from the apparent fire promotion, shares the life form traits listed here as fire-ephemeral characteristics. These traits are probable adaptations that, in general, allow *Muehlenbeckia* species to survive, and be promoted by, disturbance such as occurs on forest margins, in landslides and along road verges (Henty 1978; Wilson 1990; Brandbyge 1992). It is likely that *M. costata* is phylogenetically predisposed to growing on rocky outcrops and is promoted by disturbance. Granitic outcrop communities are potentially some of the least disturbed and fire, although relatively infrequent, is the only large disturbance apart from possibly severe drought, with any regularity. There is a need for phylogenetic analyses that would include all assessments of what are true adaptations.

The germination responses after fire are likely more a consequence of fire being a disturbance factor than to species evolving fire specific strategies due to a fire prone environment. Fires provide open high light conditions and extremely low competition sites with abundant free nutrients especially when they occur in communities intolerant of fire such as on granitic outcrops. The species studied here, in addition to fire-ephemeral characteristics, all share other strategies. All are primarily low growing prostrate or procumbent plants with flat well-displayed and often large leaves. Fire ephemeral species on outcrops are poor competitors that are obligately restricted to high light environments (Platt 1951; Baskin & Baskin 1988; Sampson *et al.* 1988; Ware 1991; Section 1.5.4). The traits shown by these supposed fire-ephemeral species on outcrops suggest that they are competition evaders that demand extreme high light environments. The same may also be true for other plants that occur on outcrops only after disturbances other than fire (Murdy 1966).

The results presented here are based largely on observations and experiments conducted in the north eastern heath dominated communities. As such they may not be easily transferable to the western part of the Batholith that is dominated by shrublands, herbfields and grasslands (Chapter 4). Fires are less frequent in the western part of the batholith and are usually not of such a high intensity due to the lower fuel loads and less flammable vegetation. It is therefore, even less likely that fires would affect granitic outcrops in the west. In addition a number of the fire induced ephemerals discussed here are restricted to high altitudes. Fire may still be important in inducing the germination of species otherwise not found in these environments. Hunter (1998c) discusses the

changes noted after 100 years of absence of fire gleaned from historical collections and descriptions at Howell (HC) (Figure 2.1). At Howell (HC) *Monotaxis macrophylla* and *Muehlenbeckia rhyticarya* were noted at the turn of the century when fires were more common, but have not been seen since, despite almost yearly student excursions and much interest by botanists over the last 30 years. During this survey some of these taxa, particularly *Muehlenbeckia rhyticarya* and *Actinotus gibbonsii* have been seen in burned areas of less than a hectare in size at Howell (HC) and Kings Plains (KP) (Figure 2.1) resulting from lightning strikes (Hunter 1999b).

9.4.3 Germination trials and post-fire germination

Despite high viability, no germinations occurred in the *M. macrophylla* trials, and although there was more germination in smoked treatments of the *M. costata* trials this was statistically insignificant. Previous studies have shown that endemics of granite outcrops require cold treatment for effective seed germination and that aging is necessary (Chapman & Jones 1971). Roche *et al.* (1997) showed that there was a great heterogeneity in germination strategies in Australian species but that some taxa positively responded to seed aging before smoke treatments. Chapman and Jones (1971) postulated that it would be deleterious for outcrop species to germinate immediately, as there will be competition from those already established and it is likely the climate will be unfavourable in such unpredictable environments (Chapter 1). These species spread widely and dominate large areas rapidly, seeds that germinated immediately are likely to be unsuccessful. All seed of *Monotaxis* was fresh and taken from living plants. Not all *Muehlenbeckia* seed was fresh, but included a small portion of seed having been sieved from the surrounding soil. This seed bank seed alone could explain the low germination rate encountered in this trial.

Auld and Bradstock (1996) have shown in forested areas that temperatures can reach 60°C in the top 0.5 cm of soil and that this is sufficient to break the dormancy of many fire induced legumes. Such soil temperatures are frequent on outcrops even without fire (Chapter 1), yet sporadic germinations of the species discussed in the preceding sections do not occur. Twenty-six of the 29 seeds that germinated in the *M. costata* germination trial did so when treated to temperatures of over 80°C (up to 120°C).

Such anecdotal and qualitative evidence suggests that fire-induced germination of ephemeral taxa on outcrops is reliant on seeds being of sufficient age and fire temperatures being above 60°C. Implicit in this scenario is that seeds must be able to remain dormant in the seed bank for decades or even centuries and that only high temperature fires will induce germination.

9.4.4 Fire and conservation and management of granite outcrops

The frequency of large and extensive fires such as those that occurred in 1994 is low and it is, therefore, not surprising that so few collections and sightings have been made of species such as *Monotaxus macrophylla* or *Muehlenbeckia costata*. The development of management strategies for the promotion of these species is problematical. General fuel reduction burning for asset protection in the surrounding forest and woodland systems is common on the New England Batholith, particularly along the eastern escarpment. Such strategies are likely to decrease the likelihood of extreme fires that are needed to promote germination on the larger outcrops. Direct ignition of outcrops is labour intensive and the required combination of environmental factors for promoting these fire-ephemeral species is unknown. Certainly fires of the wrong intensity or at the wrong time of the year could be harmful (Bradstock & Auld 1995). The experimental results suggest that the responses of patches of vegetation on outcrops are highly individualistic and framed by the initial species composition and source pools both in the seed bank and as above ground extant individuals. These communities, particularly on high altitude granitic outcrops of the New England Batholith, have evolved with a frequency of fire that is much reduced compared with the surrounding vegetated matrix. Also the recorded history of the district does not shed much light on past fire frequency. Therefore, an increase in the frequency of fires on outcrops may have undesirable effects on outcrop communities, particularly as the majority of species may be fire evaders (Figure 9.14). Thus, fire management for conservation of granitic outcrop floras is problematical and will need separate consideration from fire management regimes for the region in general.

Fire is a natural component of many communities within Australia. Much research has been conducted into the effects of fire regimes (frequency, intensity and seasonality) on individual species and on communities as a whole. The research is often habitat and site specific and the usefulness of the findings to other areas (even somewhat similar ones) is debatable. Outcrop communities are generally collectively lumped with other shrubland or heath communities although this allocation is often inappropriate from a management perspective. For instance 'heaths' in general are thought to regenerate well after fire, having a high proportion of resprouting species. Russell and Parsons (1978) showed that 73% of shrub species in 'heaths' at Wilson's Promontory were able to regenerate from resprouting. These researchers showed that fire intervals of 10 years were likely to cause only minor changes in floristics and that an inter-fire periods greater than this would cause a decline in species richness. Similar results have been achieved in heaths in coastal areas of New South Wales (Cary & Morrison 1995). The granitic outcrop 'heaths' do not respond in the same way as other structurally similar communities. Even where the same taxa are shared with communities their responses may be different. Observations of species responses to fire may be site specific (Benwell 1998). Lawler *et al.* (1998) provides, evidence to suggest that there is a decreased ability of *Eucalyptus mitchelliana* to respond to fires on exposed granite sites compared with nearby stands away from rocks. This does not mean that fire should be permanently excluded from outcrops but that the inter-fire periods should be on a much longer time scale than for surrounding heathlands.

Much fire research emphasises species richness as a management goal. In most situations, site richness (alpha diversity) is achieved by maintaining communities at an intermediate stage of development by constant but moderate disturbance. Variability in fire regimes is the goal suggested by recent research (Bradstock *et al.* 1995; Conroy 1996). Gill (1977) and Bradstock *et al.* (1995) propose that mature systems should remain, even though richness will decline at the site level (alpha diversity), richness will increase between habitats (beta diversity) thus increasing at the landscape level (gamma diversity).

9.4.5 Conclusion and areas for further research

The majority of the species on granitic outcrops are fire evaders, even in areas where the surrounding vegetation has evolved under regimes of consistently high frequency fire. Most granitic outcrop taxa are obligate seeders and may not encounter fire for many decades or even centuries. Granite outcrop taxa are competition evaders and require high light environments (Chapter 1). Once individuals occupy spaces new seedlings may no longer be able to compete with those already established as light may become limiting. This may explain why in mature vegetation patches on rock outcrops germination of new individuals is rarely seen. Moran and Hopper (1983) showed that *Eucalyptus caesia* had virtually no seedling recruitment in the absence of fire on outcrops, even though fire may not occur for centuries. Disturbances such as fires may be important to granitic outcrop communities even though they occur very infrequently. They are likely to provide chances for disturbance induced taxa to grow and replenish seed banks, and chances for the more common taxa to regenerate (Ibisch *et al.* 1995). The fire-ephemeral flora is only present on some outcrops. These species are likely to have a long lived seed bank requiring soil temperatures above 60°C and aging to break dormancy.

Many aspects of the dynamics of granitic outcrop vegetation have been shown to have inherent variability (Chapter 1, 4, 5 & 7). The fire trials conducted here show the large degree of variability in response to a disturbance such as fire or clearing at a local level (Section 9.3.3). Qualitative evidence collected during the survey shows that a fire induced flora exists but that it also may only occur on some outcrops and that it requires specific cues before appearing (9.3.1). Such inherent variability is a landscape feature that is framed by the 'island-like' nature and by the small populations found on outcrops. These features enable a high level of richness and diversity to be maintained across the landscape (gamma diversity). The ephemeral disturbance induced flora helps increase and maintain species richness and diversity spatially and temporally (beta diversity) on a landscape scale (gamma diversity). Results from Chapter 3 also show that this ephemeral flora can increase the effective insularity (QI score) of an outcrop. Ibisch *et al.* (1995) suggests that outcrop communities belong to a syndynamic complex

in which total diversity is maintained by progressive vegetation change and degradation caused by natural disturbances.

The trials discussed here are small, restricted to one region over only one year, and to a small number of taxa. Larger trials are needed both within individual regions and across the batholith at differing intensities and seasons. One fire may determine the response of a plant but consideration of numbers of fires in an area over time are necessary to decide the responses of assemblages of species (Williams & Gill 1995). Germination trials may need to take into account seed age as well as other factors such as cold treatments. Such experiments are of importance as diversity and richness is maintained temporally and spatially by disturbances, of which fire, is probably the most common within the New England Batholith. The 'island-like' nature of this system increases the stochasticity of derived responses due to variable species composition induced by small population sizes and the fact that any disturbance is unlikely to be encountered by all 'islands' in the same 'archipelago'. Experiments will need to be modified to take into account the comparative larger temporal dimension where generation times may be in the hundreds or even thousands of years (Moran & Hopper 1983). An important adjunct to the research conducted within this chapter would be investigations into the individual responses of species to the experimental burning procedures.