Influence of fire severity on fynbos plant communities and mode of regeneration in the southern Cape Peninsula

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Abstract

Fire severity influences species diversity and alters species habitat requirements, therefore it is important to understand what effect severity has on fynbos communities. The smallest twig diameter of burnt skeletons was used as a severity index and compared between two fire events for 9 permanently marked plots on the southern Cape Peninsula. Species diversity, richness and composition were analysed for one year and eight years after a fire event in 2000 and 3 years after a fire event in 2008. Twig diameters for *Erica*, *Leucadendron*, *Rhus* and *Acacia* species were significantly greater for the 2008 fire than the 2000 fire event, indicating that the fire was more severe in 2008. The ratio of reseeders to resprouters changed from 8:2 in 2001 and 2008 to 7:3 in 2011 post the second fire event. Thus resprouters were not negatively affected by increased fire severity. Ericoid reseeders appeared to favour more intense fires. Succession of fynbos communities had a greater influence on species diversity, richness and composition than fire severity. Thus fire severity does not influence fynbos communities and factors that influence species survival between fire events, such as succession or alien plant invasions, may be more influential than fire severity.

1. Introduction

The Cape Peninsula falls within the Cape Floristic Region and is dominated by fire-driven fynbos shrublands. Fire frequency, intensity, extent and seasonality influence species diversity and co-existence of fynbos communities (Bond and Keeley, 2005). Changes in the fire regime alter species habitat requirements resulting in shifts of colonizing species (Bond and van Wilgen, 1996; Thuiller *et al.*, 2007). Understanding shifts in fynbos communities will help with conservation strategies and management of fire regimes across the Cape Floristic Region (Midgley *et al.*, 2003). However, it is difficult to tease apart which components of a fire regime result in changes within plant communities.

Fire severity is a proxy for fire impact on different ecological components and assists in explaining plant community shifts as an integrated measure of the fire regime. Various factors influence the severity of a burn; such as wind speed, fire intensity, season, moisture, vegetation type and biomass load (Bond and van Wilgen, 1996). These factors promote variable fire severities thereby maintaining heterogeneity in the landscape (Bond and van Wilgen, 1996).Thus it is important to select a severity index that detects a range of intensities and reflects the ecological impacts of the fire (Euston-Brown *et al.*, 2002).

Euston-Brown *et al.*, (2002) suggests that the smallest twig diameter post-fire is a good proxy for determining fire severity in fynbos communities. Smallest twig diameter from burnt skeletons has successfully been used as a severity index in other Mediterranean type ecosystems such as Californian chaparral (Moreno and Oechel, 1989; Keeley, 1998). The index is based on the assumption that smaller diameters of remaining twigs would indicate less severe fires than larger diameters. However, the effect of fire severity varies with different species susceptibility to burning; therefore the index can only be compared between the same species or closely related species (Euston-Brown *et al.*, 2002).

Fynbos species richness and composition have different stages of succession at different ages post burning (Hoffman *et al.*, 1987; Bond and van Wilgen, 1996; Rutherford *et al.*, 2011). Some studies show that species richness starts declining shortly after the first year post-fire (Privett *et al.*, 2001) and others show that species richness increases until three years post-fire (Hoffman *et al.*, 1987; Rutherford *et al.*, 2011). By the third year post-fire most short-lived reseeders and under-canopy vegetation start disappearing due to shading and competition as longer-lived reseeders and resprouters age and dominate the canopy cover (Kruger, 1983; Le Maitre and Midgley, 1992; van Wilgen and Forsyth, 1992). Therefore, when making comparisons between different plant communities the age of the fynbos being sampled is important.

Fire intensity may influence post-fire vegetation recovery through alteration of soil properties, reduction of soil seed bank and loss of resprouting organs (Keeley, 1998). Species can regenerate post-fire via deep rooted underground storage organs such as lignotubers, corms and bulbs, or through sprouting from shallow rooted woody tissue exposed at the soil surface (Linder and Ellis, 1990). Resprouting species that regenerate through shallow rooted sprouting are most vulnerable in high severity fires (Keeley, 1998). For example, Californian chaparral resprouters were found to decrease in cover after severe fires (Keeley, 1998). Numerous fynbos species are reliant on soil stored seed banks for regeneration, thus severe fires which scorch the soil may negatively influence recruitment of obligate seeders (Bond *et al.*, 1999). Scorching of soil was found to be a particular problem in alien invaded areas and in patches where alien invasive plants were stack piled (Euston-Brown *et al.*, 2002). Although seeders are able to quickly re-establish within these heat scarred areas from surrounding vegetation, obligate resprouters which have a lower rate of sexual reproduction are more likely to be lost from communities after severe fires (Bond and Wilgen, 1996; Buhk *et al.*, 2007).

1.1 Background and Objectives

In January 2000, the Cape Peninsula experienced one of the most intense wild fires, which burnt about 8000 hectors of pristine fynbos and alien invaded areas. In November 2000, Euston-Brown *et al.* (2002) set up 52 permanent plots in these burnt areas and studied the impact of different alien clearing treatments on fire severity and its influence on post-fire pant recruitment. A re-survey was done by Blanchard *et al.* (2011, unpublished) in January 2008, which determined the impact of the different clearing treatments on post-fire succession and vegetation recovery. Shortly after the re-survey a fire re-burnt 11 plots in February 2008. Fire severity measurements were sampled within a month post fire by Ryan Blanchard and Doug Euston-Brown. In June 2011, a fourth survey for 9 of the re-burnt plots was done and the data collected over the past 11 years were analysed for the 9 plots sampled. Detailed records of how species composition and regeneration modes change from one fire to the following fire are rare, making this study important for understanding the influence of fire regimes on fynbos communities.

The objective of this study was to determine how species diversity, richness and composition have changed over the past 11 years and whether changes are linked to severity of the two fire events. Differing fire severities may preference different regeneration modes and therefore different species. Thus the study also determined how fire severity impacts on the ratio of reseeders to resprouters. It was expected that if the fire severity index for the two fires are similar, then changes in fynbos communities and regeneration modes would be due to alternative driving forces, such as vegetation succession or alien invasive plants. If dramatic differences in fire severity occur then changes in communities and regeneration modes may be due to this influence.

2. Methods

2.1 Study Sites

The study sites were located between Kommetjie, Scarborough and Simonstown on the southern part of the Cape Peninsula (Figure 1). Only burnt sites were re-sampled and the two wetland sites were excluded from the analysis. Any patches which did not burn within sites were also excluded. Sites not invaded by alien plants were designated as pristine and sites which currently have alien plants or have been cleared of alien plants since the 2000 fires were designated as alien sites (Table 1). For further detailed description of site choice and characteristics see Euston-Brown *et al.* (2002).

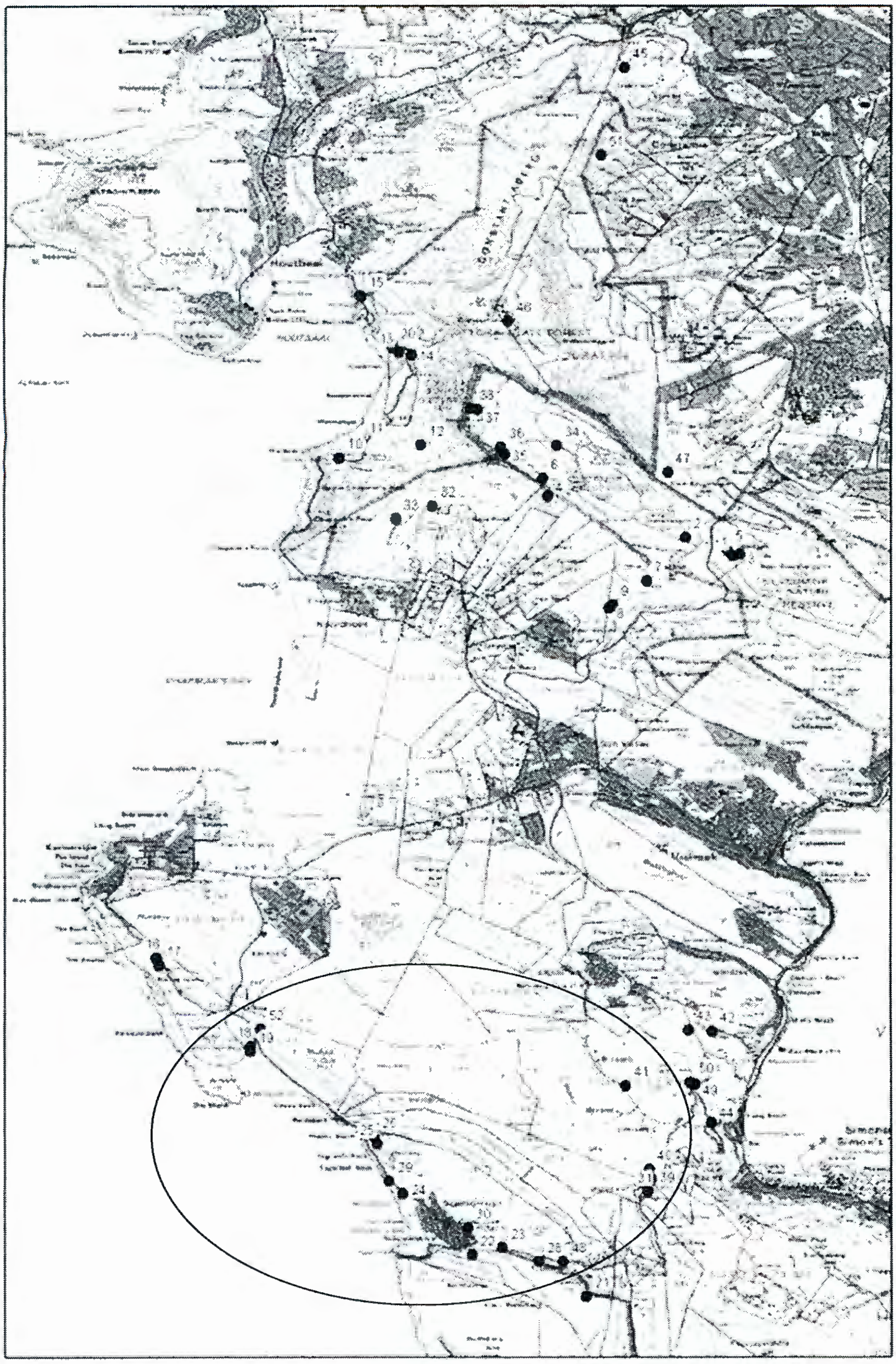

Figure 1. Map of the southern Cape Peninsula showing the original sampled plots by Euston-Brown *et al.* (2002) with current sample area encircled (taken from Blanchard *et al.* 2011, unpublished).

Table 1. The habitat, alien status and number of each site sampled.

Habitat	**Aliens**	**No. of Sites**	**Site Numbers**
Table Mountain Sandstone/Gravel fynbos	Absent (Pristine)	3	40,23,30
	Present (Alien)	6	25, 28,39,41,48,52

2.2 Sampling method

The data collection followed methods used by Euston-Brown *et al.* (2002) and Blanchard *et al.* (2011, unpublished), to ensure research comparability. A full description of sampling methods can be found in Euston-Brown *et al.* (2002). Each site covered a 20x30 meter area, which was divided into six 10x10 meter plots and two 1x1 meter sub-plots nested on opposite corners of each 10x10 meter plot to represent different scales. Although, nested plots present problems with pseudo-replication, independence is unimportant when taking means across multiple plots for each scale (Keeley *et al.*, 2003).

Sampling of vegetative variables were done in 2001, 2008 (pre-fire) and 2011. All species within the first 100 m^2 plot were identified to the lowest taxonomic rank and only new species within the remaining 5 plots were recorded. Estimates of percentage cover and height for different growth forms as well as regeneration modes were made in each 100 m^2 plot. Detailed sampling was done in the 1m^2 sub-plots by including data on height, abundance, regeneration mode and the number of new seedlings for each species. Using the data base developed by Euston-Brown *et al.* (2002) for different plant functional types, deep rooted (underground storage organs) and shallow rooted (sprouting from woody tissue) resprouting species were determined.

Fire severity was measured in 2001 and 2008 from post-fire skeletal remains to estimate the severity of the burn. The smallest twig diameter was measured below and above 0.5 meters from the ground for two different plants within each 1m^2 sub-plot. It was assumed that smaller twigs will be left after less intense fires versus higher intensity burns. Four different genera were used to represent different growth forms for comparing severity between the two fire events. *Acacia, Rhus, Erica* and

Leucadendron represented alien trees, indigenous trees, ericoids and proteoids respectively. These genera were selected because they consisted of at least 3 twig measurements for each year over all the sites sampled.

2.3 Data Analysis

Shannon-Wiener, Equitability and Simpsons diversity indexes were calculated at the 1m^2 scale for each individual sub-plot. A one-way analysis of variance (ANOVA) was calculated between the three survey events in the program STATISTICA 2010. Thereafter, if the ANOVA was significant a Tukey HSD post hoc test was done.

Species richness was determined for each scale, where each individual sub-plot was used for the 1m^2 scale and only the first 10x10m plot was used for the 100m^2 scale. All species found across the 6 plots were used for the 600m^2 scale. Species richness was Log+1 transformed to meet normality assumptions before doing an ANOVA and post hoc Tukey HSD test between the three surveys. The difference in species richness between alien invaded and pristine sites was determined for each year over different scales using a student t-test with the transformed data.

Species composition was computed in PC-Ord 5.0 using a Nonmetric Multidimensional Scaling ordination (NMS) with Sørensen similarity index. Species absence/presence data was used and grouped according to year and alien status at the 600m^2 scale. Species that only occurred once over all sites were excluded from the ordination to reduce the influence of rare species.

The twig diameters were Log+1 transformed to meet assumptions of normality prior to student t-test analysis between 2000 and 2008 fires. T-tests were carried out for *Acacia, Rhus, Erica* and *Leucadendron* twig diameters. Zeros were removed from the severity data, because it could either be an indication of high severity or represent vegetation with high flammability, such as graminoids.

The percentage cover of resprouters versus reseeders were graphed and compared across the three vegetative surveys using a Kruskal-Wallis non-parametric ANOVA at the 100 m^2 scale. The proportion of the total number of reseeders to resprouters for all sites was determined for each year. The abundance of dominant reseeder and resprouter genera were summed and compared between the three surveys. Furthermore, changes in abundance of different resprouting modes (deep or shallow rooted) were compared between the surveys.

3. Results

3.1 Species diversity, richness and composition between the different surveys

Species diversity in 2008 was significantly different from 2001 and 2011 surveys according to both Shannon-Wiener and Simpsons Indexes (Table 2). The Equitability Index was not different between surveys.

Table 2. The mean (±standard deviation) for the different diversity indexes and results from one-way ANOVA and Tukey post hoc test at the 1m^2 scale.

Diversity Index	**2001**	**2008**	**2011**	**Df**	**F**	**P**
Shannon-Wiener	1.97 (0.70)^a	1.67 (0.55)^b	1.99 (0.54)^a	309	9.469	<0.001
Equitability	0.84 (0.10)^a	0.83 (0.11)^a	0.85 (0.09)^a	300	1.120	>0.05
Simpsons Index	6.63 (3.59)^a	4.79 (2.28)^b	6.64 (3.39)^a	309	12.131	<0.001

Species richness increased as the sample area increased (Figure 2). Species richness in 2001 at the 600m^2 scale was significantly greater than species richness in 2008 and 2011 (ANOVA: df=30, F= 11.97, p<0.01).

The greatest difference in species richness between pristine and alien sites was seen at the 600m^2 scale in 2001 (Ttest 2001: df=122, F=1.56, p<0.001) and the least difference at the 1m^2 scale (Figure 3). At the 600m^2 scale the species richness decreases more dramatically from 1 year to 3 years post fire compared to loss of species richness from 3 years to 8 years post fire, despite the shorter time period. The difference in species richness is trivial (<10 species) between pristine and alien sites in 8 year old vegetation for all scales (Figure 3).

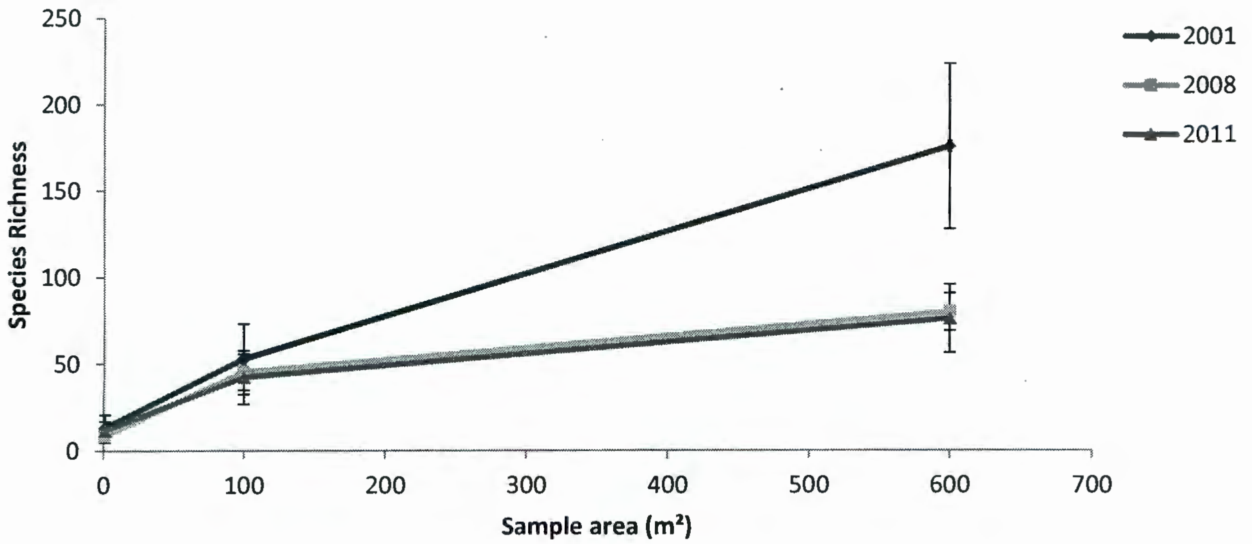

Figure 2.The mean species richness with standard deviation for three surveys grouped into different scales (m^2).

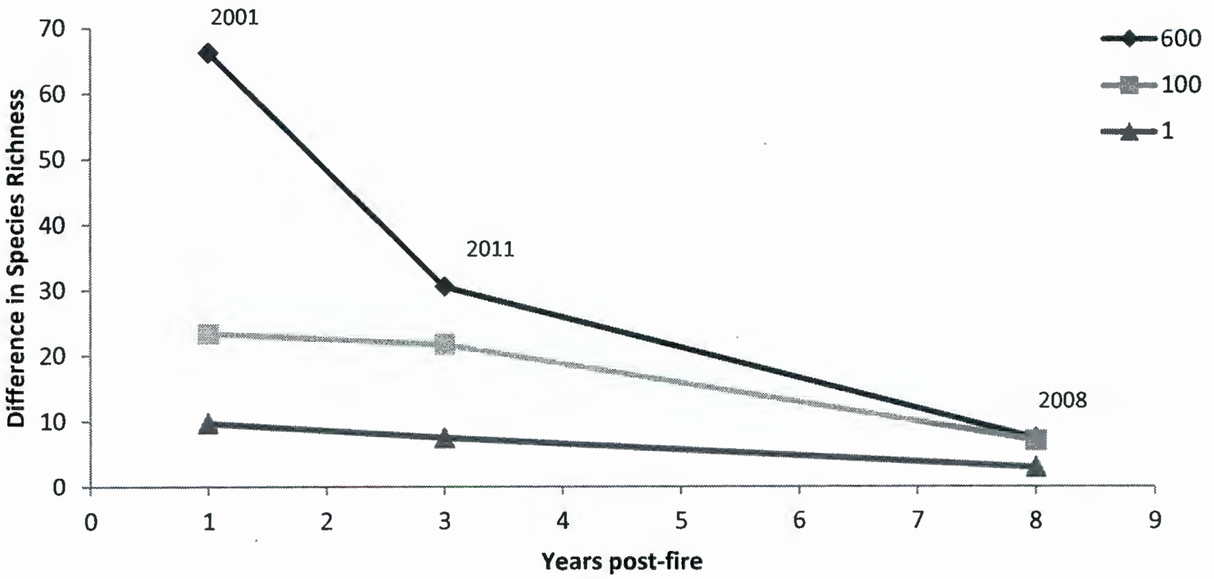

Figure 3.The difference in mean species richness between pristine and alien fynbos sites over age of vegetation post-fire for each scale (m^2).

Results from the Nonmetric Multidimensional Scaling (NMS) ordination are shown in Figure 4 for each sampled site. Two distinct groupings can be seen between the different survey years. Sites surveyed in 2008 and 2011 were all distributed closer to axis 2, compared to the sites in the 2001 survey which were distributed further from axis 2 (Figure 4). The distance between sites in 2001 and 2008 was greater than the distance between sites in 2008 and 2011.

Sites were also grouped as 'pristine', which did not have alien species. Pristine sites were distributed on the bottom left quadrat and were separated from the alien invaded sites which were distributed towards the top right quadrat of the ordination (Figure 4). Site 39 was more closely distributed relative to the pristine site grouping and sites 52, 41 and 28 became more similar over time.

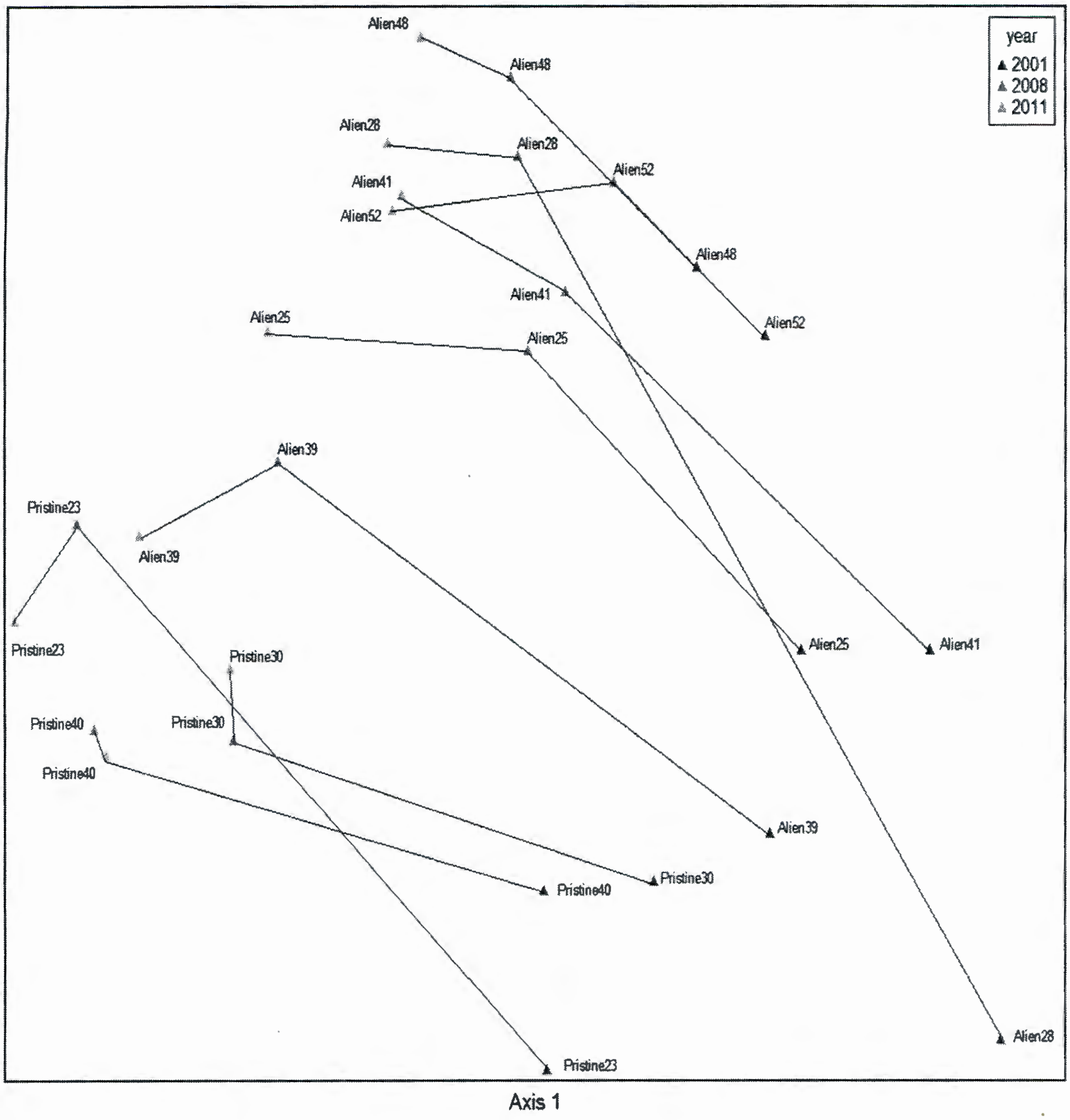

Figure 4. NMS ordination of species composition (absence/presence) at the 600 meter scale grouped by year of survey (500 iterations with 27 plots and 344 species; stress=12.08; stability=0.006; p=0.004).

3.2. Fire severity between the two fire events

Alien invaded sites 25 and 48 were burnt to the ground in 2001, thus no twig diameters were recorded. *Erica* (Ttest: df=56, F=1.48, p<0.001) and *Leucadendron* (Ttest: df=34, F=5.4, P<0.001) showed twig sizes that were significantly larger for 2008 than 2001 below 0.5 meters from the ground (Figure 5). *Acacia* (Ttest: df=31, F=1.56, p>0.05) and *Rhus* (Ttest: df=12, F=3.27, P>0.05) twig diameters were not significantly different between the two fires below 0.5 meters from the ground (Figure 5). *Acacia* (Ttest: df=10, F=43.42, p<0.001), *Erica* (Ttest: df=15, F=4.21, p<0.001), *Leucadendron* (Ttest: df=34, F=31.48, p<0.001) and *Rhus* (Ttest: df=7, F=1.28, p<0.001) showed significantly larger twig sizes in 2008 than 2001 above 0.5 meters from the ground (Figure 6).

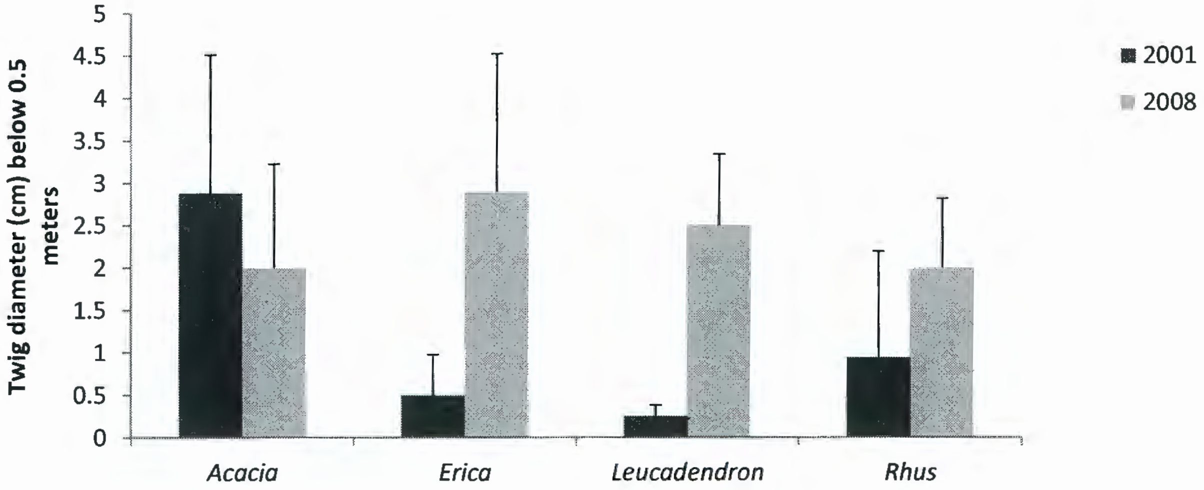

Figure 5. The mean and standard deviation of twig diameter (cm) below 0.5 meters from the ground for different growth types following post-fire severity surveys in 2001 and 2008.

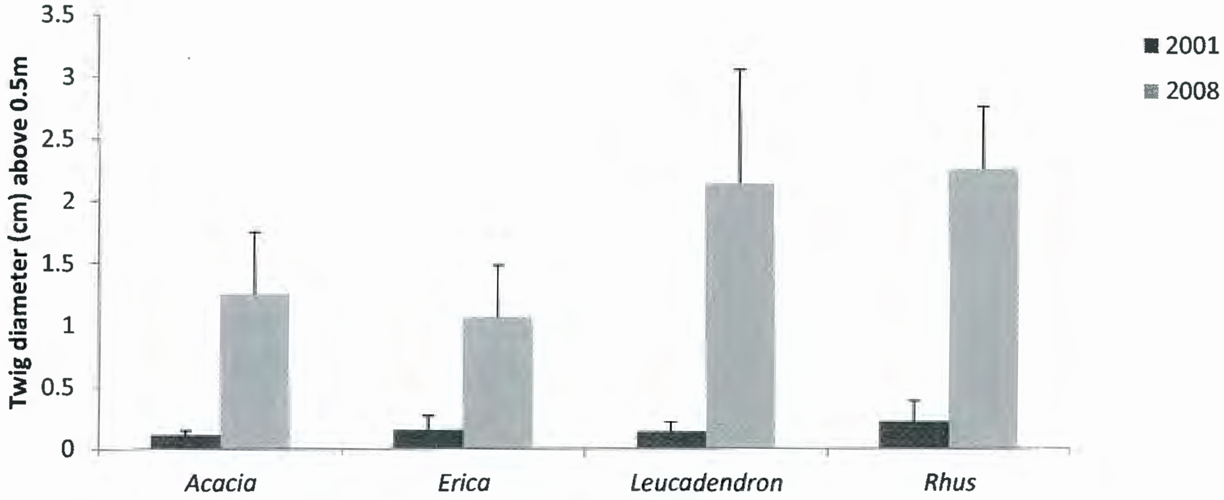

Figure 6. The mean and standard deviation of twig diameter (cm) above 0.5 meters from the ground for different growth types following post-fire severity surveys in 2001 and 2008.

3.3. Reseeder and resprouter representation

As vegetation age increased, the percentage cover of reseeders increased (Figure 7). The percentage cover reseeders in 2001 was significantly lower than in 2008 and 2011 (Kruskal-Wallis: N=162, H=84.17, p<0.001). The percentage cover of resprouters in 2011 was significantly greater than in 2001 and 2008 (Kruskal-Wallis: N=162, H=20.00, p<0.001). The proportion of total number of reseeders to resprouters changed from 8:2 for 2001 and 2008 to 7:3 for 2011. Despite difference in plot cover between reseeders and resprouters increasing to 63% in 2008 (8 years post fire), the proportion of reseeders to resprouters was not different from 2001 (Figure 7).

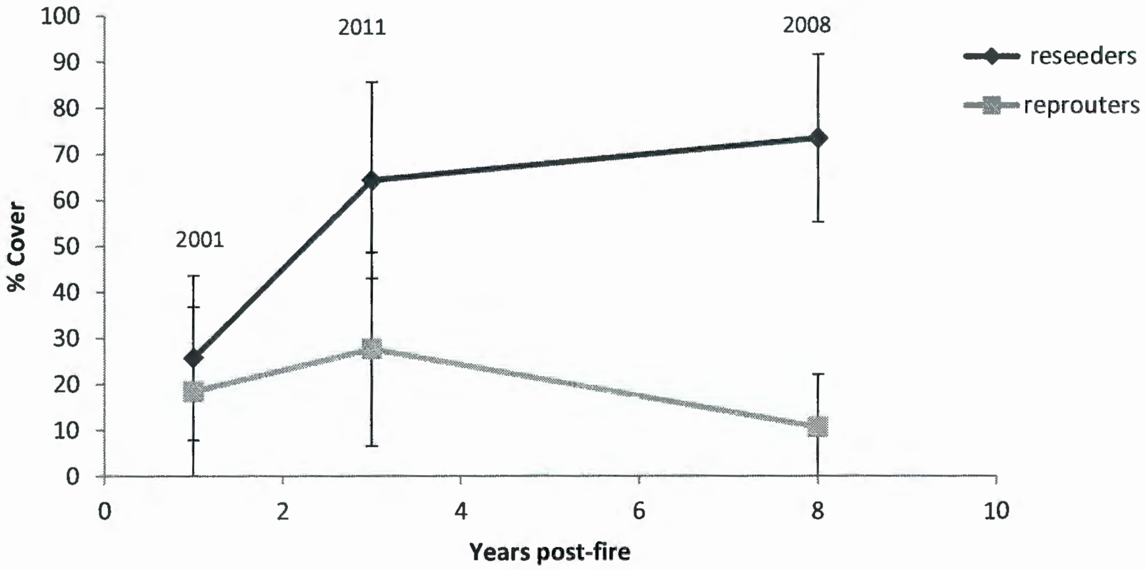

Figure 7. The mean percentage cover and standard deviation for reseeders and resprouters over post-fire age at the 100 m^2 scale.

Resprouting species within the genera *Acacia, Elegia, Pseudopentameris* and *Thamnochortus* increased in abundance post 2008 fire compared to 2001 and 2008 (Table 3). However, species within the genera *Ficinia, Olea,* and *Pelargonium* decreased in abundance since the 2008 fire. *Rhus, Pelargonium* and *Olea* had a number of mortalities between the 2001 and 2008 time period. Resprouting *Ehrharta* and *Tetraria* species appear to decrease in abundance as vegetation age increased (Table 3).

Reseeding species from the genera *Acacia, Elegia, Erica, Leucadendron, Metalasia* and *Passerina* all increased in abundance post 2008 fire (Table 3). A large number of *Pelargonium* mortalities were noted between 2001 and 2008 as well as a decrease in abundance from 2008 to 2011. *Aspalathus,* *Ficinia, Othonna* and *Roella* decrease in abundance as vegetation age post fire increased. Reseeding *Ehrharta* species show the opposite trend and increased in abundance as vegetation age post fire increased (Table 3).

Table 3. Total abundance of dominant resprouting and reseeding species grouped into genera for 1m^2 sub-plots over the three sampling years. New seedlings of resprouting species were noted as seeders.

Resprouting	**Sprouting mode**	**2001**	**2008**	**2011**	**Reseeding**	**2001**	**2008**	**2011**
Acacia	shallow	0	2	9	*Acacia*	4	14	29
Ehrharta	deep	62	0	7	*Aspalathus*	276	23	110
Elegia	deep	5	5	14	*Ehrharta*	40	117	46
Ficinia	deep	333	93	78	*Elegia*	5	11	21
Olea	shallow	34	11	6	*Erica*	37	132	176
Pelargonium	deep	26	11	10	*Ficinia*	317	146	194
Pseudopentameris	shallow	5	32	39	*Leucadendron*	82	56	86
Rhus	shallow	22	7	7	*Metalasia*	63	47	85
Thamnochortus	deep	27	75	81	*Othonna*	38	5	33
Tetraria	deep	84	19	57	*Passerina*	55	49	74
					Pelargonium	302	22	14
					Roella	670	5	209

4. Discussion

Understanding the influence of fire severity on fynbos communities and mode of regeneration will assist in improved management of the fire regime. There are a number of papers on how fire regime components influence fynbos communities, however most focus has been on fire frequency (Bond and van Wilgen, 1996). This research is an extension of the first work done by Euston-Brown *et al.*, (2002) on fire severity using a post-fire index. It is also one of few studies that consist of detailed changes in community structure and composition over two consecutive fires in permanently marked plots.

The burn in 2008 was significantly more severe than the burn in 2001 according to the smallest twig diameter above 0.5 meters for all four genera of plants (Figure 6). Remaining twigs below 0.5 meters were not different between the two burns for *Acacia* and *Rhus* species, possibly because they do not burn as severely as *Erica* and *Leucadendron* species or because they were taller and just did not burn below 0.5 meters.

Using species of the same growth form ensures that fire severity is measured without the influence of different species flammability. However, using a severity index specific to genera/species is limiting because they do not occur over all sampled plots. Thus, this index was unable to provide a standardized indication of severity across all plots sampled. Therefore, the comparison of severity between fires only includes a subset of sites. In future, sampling of twig diameters at the 100 m^2 or 600m^2 scale instead of at the 1m^2 scale may increase the likelihood of comparing the same genera across more sites. Results from the fire severity index could also be improved by using an index applicable across all sampled plots and not limited by variability of species flammability.

The species diversity in 2008 was significantly lower than in 2001 and 2011, which is expected in fynbos communities as shorter lived species start dying off with increasing stand age (Table 2; Kruger, 1983; Le Maitre and Midgley, 1992; van Wilgen and Forsyth, 1992). The equitability index was similar for all three surveys, indicating that there was a similar distribution of species abundance (Table 2). Species diversity and species richness was not different between 2001 and 2011 at the 1m^2 scale (Table 2 and Figure 1). However, species richness was significantly greater at the 600m^2 scale for 2001 than 2011. Thus species diversity may be different between 2001 and 2011 if abundance of species was collected and diversity calculated at the 600m^2 scale.

The 2001 peak in species richness was due to the large number of species found in pristine plots (Figure 3). This high species richness can be attributed to post-fire annuals and reseeders, which die-off one to three years post-fire, explaining the lower species richness in 2011 (Bond and van Wilgen, 1996; Rutherford *et al.*, 2011). There was little difference in the number of species found between alien and pristine plots at the 1m^2 scale (Figure 3). However, species richness was severely impaired in alien sites at the 100 m^2 and 600 m^2 scales (Figure 3). This difference between micro- and macro-scales becomes trivial in older vegetation as species richness becomes depleted in pristine sites, as was seen in 2008 (Figure 3).

There are a few explanations for both the loss of species in alien sites (Figure 3) and the tendency of alien plots to become more similar in species composition over time (Figure 4: plots 28, 41 and 52).

Increased fire intensity from high alien biomass and heat scars from stack piling may result in loss of resprouters and seeds in soils (Euston-Brown *et al.*, 2002). The physiological functioning of alien plants may have an effect on the soil chemistry and nutrient composition influencing species recruitment ability and survival (Morris *et al.*, 2011). Furthermore, alien species may also establish before indigenous species or out-compete them in the post-fire environment (Morris *et al.*, 2011).

The separation of species composition into pristine sites and alien sites was not unexpected, because of the many more species found in the pristine sites (Figure 4). Interestingly, site 39 was more similar to pristine than alien in 2011 (Figure 4). This indicates that the site recovered from the invasion of alien plants, possibly because it was surrounded by pristine fynbos resulting in a faster recovery time compared to other cleared alien sites.

It was expected that the community composition in 2001 would be more similar to the 2008 survey, considering that a fire occurred between the 2008 and 2011 surveys. The NMS ordination of fynbos community composition however, showed that sites sampled in 2008 were more similar to the 2011 survey than to the 2001 survey (Figure 4). This result highlights a few important points. Firstly, not having fire present in a fynbos community is more of a disturbance to the species composition than having a fire burn all the vegetation. Secondly, the severity of the burn experienced did not have a large influence on the community composition as evidenced by the similarity of 2008 and 2011 plots. Lastly, changes in species composition prior to the fire event may determine the species composition post fire. However this point cannot be made with certainty, due to the sampling of different stages of vegetative succession. Another problem with the ordination results is the similarity seen between 2008 and 2011 surveys which may be a result of the species richness being similar. In other words the 2001 species richness being significantly greater than 2008 and 2011 (Figure 2) thus forcing them to be more similar to each other.

The results show that species composition was stable after a fire event within and across sampled sites, but large differences occurred between fire events. Thuiller *et al.* (2007) found high stability of species composition at the meta-scale (over all sites sampled) but high temporal turnover at the local scale (within sites). However, Thuiller *et al.* (2007) was unable to analyse independent fire events to determine the reason for the changing species composition at the local scale. Their conclusion was that variability at the local scale was due to uneven fire regime effects, post-fire environmental factors and succession within fynbos communities. However, fire regime components involved with the severity of fire actually have less influence on species composition compared to post-fire variables, such as senescence of species. Furthermore, it would have been interesting to see where plots would have distributed (closer to 2001 or 2008), if they had been sampled one year post the 2008 fire as well.

Fire severity may influence the representation of different regeneration modes within communities. The ratio of reseeders to resprouters remained 8:2 from 2001 to 2008, despite the large changes in species richness, diversity, composition and differences in percentage reseeder and resprouter cover (Table 1, Figure 2, 4 and 6). However, after the 2008 fire the ratio changed to 7:3. The percentage cover of resprouting species was also significantly greater in 2011 than in 2001 and 2008 (Kruskal-Wallis: N=162, H=20.00, $p<0.001$). Thus the more severe fire promoted a greater abundance and cover of resprouting species relative to reseeding species, which was opposite to what was expected (see Keeley, 1998).

Erica, Metalasia and *Passerina* species showed a marked increase in abundance after the more intense 2008 fire compared to the 2000 fire (Table 3). Euston-Brown *et al.*, (2002) predicted that severe burns with a smallest twig severity index of 1.5cm for ericoid species would result in high seed mortality. The *Erica* twig diameters for the 2008 burn varied from between 0.5 cm to 1.5 cm indicating a relatively severe fire (Figure 5). However, the abundance of *Erica* seeds was much higher in the 2011 survey compared to 2001 and 2008 (Table 3). This indicates that reseeding ericoid species are able to tolerate high severity fires and be successful. However the ability to survive high severe fires may be restricted to the specific fire regime of pristine sites, as *Erica* species were not found in heavily invaded sites.

The increase in reprouting species was due to the increase in abundance of *Acacia, Elegia, Pseudopentameris* and *Thamnochortus,* indicating that these resprouting species are not negatively influenced and might even be favoured under more severe fire regimes (Table 3). Bellingham and Sparrow (2000) suggested that deeper resprouters would be more abundant in areas with severe and frequent disturbances, compared to shallow rooted sprouters which would prefer less intense disturbances. Both *Elegia* and *Thamnochortus* are deep rooted species, explaining why they were less influenced by a higher severity fire. However rooting depth cannot explain why deep rooted resprouts of *Pelargonium*, *Tetraria* and *Ehrharta* decreased in abundance or why shallow rooted *Pseudopentameris* increased in abundance (Table 3). Furthermore, a greater mortality of shallow rooted *Rhus* and *Olea* species was seen between 2001 and 2008, than after the 2008 fire (Table 3). Therefore, abundance of resprouts cannot be predicted by fire severity alone and other environmental factors need to be considered such as species susceptibility to drought, senescence and the influence of alien invasives.

5. Conclusions

This study shows that fire severity has limited influence in determining species diversity, richness and composition. However other environmental factors, such as post-fire succession, alien invasion, and plant functional traits play a more important role. Reseeding ericoid species increased in abundance after the more severe fire in 2008. However, due to the limited severity index this can only be claimed true in uninvaded pristine sites. The ratio of resprouters increased relative to reseeders in the more intense fire, but there was mixed results regarding the abundance of deep and shallow rooted resprouters between the two fire events. Thus the different fire severities did not preference either resprouting mode. Although, fire severity did not drastically influence the fynbos communities, it is unknown how communities may react to more severe fires than the two fire events observed. Continuing this long term research would provide further valuable information on changes in the landscape as well as influences of current fire regimes on future fynbos communities.

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Ceramium but their increase in number is small relative to the increase in *Ceramium* biomass. Thus, once *Ceramium* populations do become large, isopods are useless as control agents.

Isopods are most valuable in combating *Ceramium* during harsh environmental conditions and when *Ceramium* populations are low. Their importance may be in retarding the initial spread of *Ceramium*, and keeping the *Gracilaria* relatively epiphyte free for a longer period of time.

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