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The effectiveness of active restoration following alien clearance in fynbos riparian zones and resilience of treatments to fire

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Abstract

In 1998, a restoration field trial was initiated in a catchment near Wellington (Western Cape, South Africa) to determine whether fynbos riparian scrub vegetation cleared of woody invasive alien trees require post-clearance restoration actions to accelerate indigenous vegetation recovery. The aim was to assess the relative effectiveness of three sowing treatments for restoring indigenous vegetation cover after the widely used "Fell & Burn" method of clearing invasive alien trees. Sowing treatments included non-invasive alien grasses to determine whether they have a negative effect on recovering native vegetation. A summer fire, eight years after trial initiation, provided an opportunity to determine how resilient restoration treatments are to alien re-invasion and fire. Restoring the site after alien clearing by sowing indigenous seeds increased both diversity, by improving species presence and abundance. However, a census done 8 years later (in 2006) revealed that seedlings of woody invasive alien plants dominated all plots, and had also survived the burn by resprouting, indicating the importance of follow-up control to justify initial clearing and restoration costs. Indigenous grass density was significantly reduced in plots where alien grasses were sown, while in the control and fynbos sowing treatment, indigenous grass density increased. By 2006, alien grass density was negligible in all treatments, indicating that the two grass species sown are not persistent or invasive. Active restoration of riparian areas after alien plant clearing has potential to facilitate vegetation recovery, but must be coupled with a long-term plan for adequate follow-up removal of post-clearance and post-fire alien recruits.

Keywords: Biological invasions; Fire; Follow-up treatment; Management; Restoration

1. Introduction

Invasive alien plants (IAPs) have detrimental effects on agriculture, forestry and human health (Wilcove et al., 1998; Walker and Steffen, 1999) and are recognized as the second-largest global threat to biodiversity after direct habitat destruction (Walker and Steffen, 1999; Latimer et al., 2004). Riverine ecosystems are naturally more susceptible to invasion by IAPs than other ecosystems (Hood and Naiman, 2000) and in South Africa, river banks and river beds are among the most densely-invaded landscape features (Richardson and Van Wilgen, 2004). This may be because riparian zones are subjected to disturbances by periodic flooding (Leopold et al., 1964; Richardson et al., 2007) as habitat disturbances have been found to promote invasion by exotic plant species (Ewel, 1986; Hobbs, 1989; Mack, 1989; D'Antonio and Meyerson, 2002; Holmes et al., 2005). Invasive alien plants may be introduced to riparian zones by human-mediated disturbance while bare sediments deposited after high flows provide an ideal environment for weed recruitment (Hood and Naiman, 2000; Foxcroft et al., 2008-this issue).

In cases where riparian zones are invaded by IAPs (mostly woody tree species in South Africa's Western Cape province), diverse riparian vegetation has been replaced by species-poor alien stands (Cowling et al., 1976; Richardson et al., 1989, 1992, 1997; Richardson and Van Wilgen, 2004). Alien plant

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invasions have thus reduced indigenous plant density and richness, causing significant changes to both above- and below-ground (seed bank) composition and guild structure (Macdonald and Richardson, 1986; Holmes, 2001; Vosse et al., 2008-this issue). In many cases, ecosystem function has been severely impaired (Cronk and Fuller, 1995; Richardson et al., 1997; Richardson and Van Wilgen, 2004).

The degree to which a riparian ecosystem recovers naturally to an indigenous state, after alien clearing, is determined by the vegetation and soil types, the alien plant species involved, the infestation age and density, the clearing treatments used, and in fire prone systems, the number of fire cycles experienced by the infestation prior to and after clearing (Macdonald, 2004). Many IAPs build up seed banks, which complicate the long-term restoration of a site and challenges management goals (D'Antonio and Meyerson, 2002). However, managers have successfully used exotic species to restore particular functions within a degraded landscape in circumstances where remaining native species are unable to deliver the desired function (D'Antonio and Meyerson, 2002).

In 1995, the South African government instigated the Working for Water programme (Van Wilgen et al., 1998) to clear watersheds of invasive trees to enhance stream flow (Le Maitre et al., 1996; Dye et al., 2001; Dye and Jarmain, 2004). Labour intensive approaches are taken with dense infestations (as was the case for our study site in 1998 (Prins, 2003)). After felling, logs are removed from the riparian zone (to prevent log-jam damage downstream), left to decompose (if slash is not too heavy), or burnt in slash piles. Ideally, native vegetation should be restored as quickly as possible because alien tree species are capable of rapidly re-colonizing the bare ground through abundant seed production, rapid seed bank accumulation and serotinous seed germination (Richardson and Cowling, 1992).

Riparian restoration is a process of re-establishing species, assemblages, structure, and/or ecological functions of the riparian habitat after alien clearing (Van Diggelen et al., 2001). If the hydrology and geomorphology of the invaded ecosystem is still functional and able to support an indigenous community, the re-establishment of species assemblages should be the target for ecosystem repair (Hobbs and Harris, 2001; Holmes et al., 2005; Richardson et al., 2007). Many riparian specialist species are relatively widespread and, in this system, are predominantly resprouters (Prins et al., 2004). Ideal situations for riparian vegetation restoration are when the site can rapidly recover through natural re-colonization from undisturbed surrounding sites (Prins et al., 2004). This saves costs on restoration and also ensures that the local gene pool is maintained (Prins et al., 2004). However, species should be re-introduced by using seed or propagated material if the surrounding landscape is highly degraded (Prins et al., 2004; Holmes et al., 2005), as was the case in our study. Prins et al. (2004) suggested that emphasis should be placed on re-introducing the common and generalist riparian shrub species and particularly species adapted to fire, as they are most likely to quickly re-establish appropriate vegetation structure and resilient plant cover.

Although the Working for Water initiative has been operational since 1995, it is not known whether cleared areas require post-clearance restoration actions to accelerate indigenous riparian vegetation recovery as little monitoring has been done (Holmes et al., 2005; but see Blanchard and Holmes, 2008-this issue). Our study made use of a "natural experiment" that was created when an unplanned summer fire (February 1998) burnt through a catchment at Oaklands near Wellington (Western Cape, South Africa), where Working for Water teams were busy clearing a dense stand of woody invasive trees. The unplanned fire provided the opportunity to monitor regeneration and erosion effects (not reported here but see Prins, 2003) of different restoration sowing treatments following the widely used "Fell & Burn" method. A subsequent unplanned fire through the site after eight years provided the opportunity to investigate the resilience of the restored treatments to alien re-invasion and fire. The study focused on a mountain stream that, prior to invasion, would likely have had a short lateral transition through wet bank, dry bank to terrestrial fynbos (Davies and Day, 1998; Boucher and Tlale, 1999). Species assemblages within these zones would have contained both characteristic riparian genera that do not have a terrestrial affinity, as well as riparian scrub species with a fynbos affiliation. We wished to test the hypothesis that sowing a mixture of riparian scrub species (e.g. *Brabejum stellatifolium*, *Berzelia lanuginosa*, *Leucadendron salicifolium*, *Rhus angustifolia*) and local terrestrial fynbos species (e.g. species of *Athanasia*, *Euryops*, *Pentaschistis*, *Protea*, *Stoebe*), is a suitable approach to initiate the restoration of mountain stream riparian structure and function where closed-stand alien stands have been tackled. We argue that once structure is in place, other riparian scrub species will re-colonize over time, provided they are in the catchment area. This is the first report of a riparian restoration trial using active species re-introduction in the Fynbos Biome. Our key questions were:

- How effective is sowing a mixture of seeds of indigenous plant species for the recovery of riparian vegetation density and cover?
- Does the sowing of non-invasive alien grasses suppress or have a negative effect on re-establishing native vegetation?
- What treatment is most effective in promoting the recruitment of indigenous species?
- How resilient are restoration treatments to alien re-invasion and fire?

2. Materials and methods

2.1. The study area

The study site is situated on Oaklands Farm at the foot of Groenberg near Wellington in the Western Cape, South Africa (33° 36′80″S; 19° 5′30″E). The vegetation type of the catchment is Boland Granite Fynbos growing in soils derived from deeply-weathered granites with the upper layers containing colluvial sandstone material (Rebelo et al., 2006). A small, alien-invaded stream, that was subject to some studies in the past (Dye et al., 2001; Dye and Jarmain, 2004), was chosen for the study. *Acacia mearnsii* De Wild. and *Eucalyptus cladocaylyx* F.Muell. had invaded the whole catchment area while *A. mearnsii* dominated the riparian zone (Prins, 2003). Between March and May of 1997, Working for Water teams clearfelled the riparian zone by felling the alien trees without removing the tree boles or slashed material. A wild fire swept through the clearfelled area on Oaklands farm early in 1998 giving rise to a clearing treatment of "Fell & Burn" within which the experiment was set up. One follow-up alien-clearing treatment was applied in January 1999 when all alien saplings were hand-pulled (carefully to avoid damaging indigenous plants). An unplanned summer fire swept through the sub-catchment in December 2005, eight years after the experiment was initiated.

2.2. Experimental design

The experimental layout consisted of twenty contiguous, 50 m^2 plots set up on the north-facing bank of the riparian zone, with the 5 m edge parallel to the stream. Each 50 m^2 plot received one of three different sowing treatments in May 1998 (hereafter named the Fynbos, Mix and Terraces treatments) or a Control treatment. These three treatments were applied to compare effectiveness in restoring riparian structure and function (specifically erosion control, reported in Prins, 2003). The Fynbos sowing treatment comprised of evenly-sown seeds of local fynbos and riparian species. The species and quantities included within the seed mix are provided in Table 1. The Mix sowing treatment comprised seeds as per the Fynbos sowing treatment with the addition of two non-invasive alien annual grass species, *Eragrostis tef* (Zucc.) Trotter and *Avena sativa* L., all evenly sown. In the Terraces treatment, the grass was separated and sown in terraces at 1m intervals (horizontally across the plot), while the Fynbos seed mix was sown between the terraces. The annual grass was added to test for increased erosion control of banks, whereas the Terraces treatment tested for a reduction in potential competition between annual grass and indigenous seedlings. The four treatments were randomly allocated to the plots, resulting in five replicates each. Under mesic conditions, *Pteridium aquilinum* subsp. *aquilinum* (Bracken) is commonly one of the first species to regenerate after fire and its dominance in these situations may hinder restoration efforts. Thus, its density was reduced by slashing prior to the initiation of the experiment. *Anthospermum aethiopicum* L. foliage was used as mulch and all plots were covered with it immediately after sowing to prevent seed loss and to retain moisture following rainfall events. The Control treatments comprised of the plots being covered with mulch only. Four 1 m^2 plots were randomly set out within each 50 m^2 plot to monitor the recruitment and survival of all growth forms.

Table 1
Species and seed volumes (e.g. no. of cones, handfuls, grams) sown per plot in Fynbos, Mix and Terrace treatments

Species	Volume
Anthospermum aethiopicum L.	*
Athanasia spp.	*
Berzelia lanuginosa (L.)	2 handfuls
Brabejum stellatifolium L.	27 fruits
Diospyros glabra (L.) De Winter	*
Euryops abrotanifolius (L.) DC.	*
Helichrysum spp.	*
Leucadendron salicifolium (Salisb.) I. J. Williams.	30 heads
Leucadendron salignum P.J. Bergius.	20 heads
Montinia spp.	*
Pentaschistis curvifolia (Schrad.) Stapf.	*
Protea laurifolia Thunb.	5 cones
Protea repens (L.) L.	14 cones
Rhus angustifolia L.	*
Stoebe spp.	*
Tribolium spp.	*
Fynbos mix (indigenous seed mix)	4 handfuls
Alien grasses	
Avena sativa L.	400 g
Eragrostis tef (Zucc.) Trotter	50 g

*Not measured, but contributed to an additional 4 handfuls of uncleaned seed per plot.

2.3. Data collection

Plots were censused 5, 6, 9, 13, 15 and 96 months after sowing although only data for the 5- (October 1998), 9- (June 1999) and 96-month (September 2006) censuses are presented. The seedling recruitment was readily monitored as plot assessments were preceded by winter rains. All seedlings were identified and classified into eight growth form guilds: ericoids, proteoids, alien forbs, indigenous forbs, alien grasses, indigenous grasses, broad-leaved shrubs and geophytes. In addition, bracken and *Acacia* and *Eucalyptus* seedlings were censused. Resprouting shoots were not scored as seedlings. Density counts were made for each guild or species and the total cover was estimated for each 1m^2 plot. Indigenous and alien species cover was also estimated during the 1998 and 1999 surveys.

A survey of all the woody plant material within the 50 m^2 plots was conducted during the winter rainy season in July 2006 before seedling recruitment to avoid trampling of emerging seedlings. All woody specimens were identified, scored as dead or resprouting and their heights estimated and basal diameters measured.

2.4. Data analysis

Data analysis focused on differences between sowing methods compared to the control over time, using repeated measures analysis of variance. Tukey HSD tests were used to determine where the differences occurred. For the final census in 2006, basal diameter of indigenous and invasive woody species was used to compare plant growth among treatments, while rank-abundance curves for woody species described evenness of species distribution and relative species dominance within each sowing treatment. In addition, 2006 size-class distributions are presented for *A. mearnsii*, *B. lanuginosa, B. stellatifolium*, *Protea repens* and *Protea laurifolia*, for each sowing treatment respectively. Size classes were calculated by multiplying the basal diameters (cm) with the estimated heights (converted to cm). Single factor ANOVA was used to test for significant differences ($P<0.05$) in basal diameter for each species recorded among the sowing treatments. Bonferroni tests were used to determine which treatments differed significantly ($P<0.05$) from each other.

3. Results

In the early stages of the study (1998, 1999), treatment effects were rarely significant although these became more significant with time. Most of the species sown in 1998 were identified on the site eight years later (2006) except for four fynbos species and the two alien grass species used in initial soil stabilization. Several species were recorded both in 1998 and 2006 that were not sown but recruited from other sources.

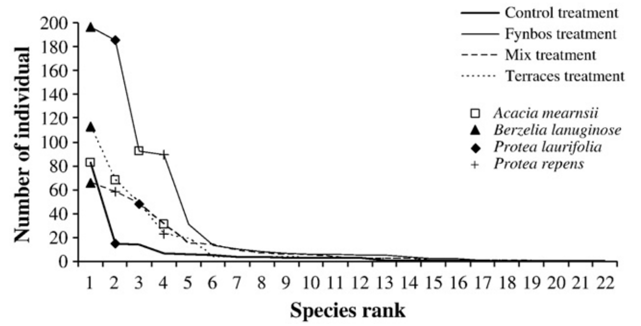

Fig. 2. Rank-abundance curves of woody species within 50 m^2 plots in Control, Fynbos, Mix and Terraces treatments measured in 2006.

3.1. Woody material

Eight years after the initiation of the experiment and after the December 2005 fire, all *B. lanuginosa*, *L. salicifolium*, *P. laurifolia* and *P. repens* adults were recorded as dead. All individuals of *Leucadendron salignum*, *Maytenus oleoides*, *Metrosideros angustifolia*, *Myrsine africana*, *Olea capensis* subsp. *capensis*, *O. europaea* subsp. *africana*, *Podalyria calyptrata*, *Rhus rehmanniana* var. *glabrata* and *R. angustifolia* and more than 90% of the *B. stellatifolium* and *Diospyros glabra* individuals present on the site were resprouting. The invasives *A. mearnsii* and *E. cladocaylyx* were still alive after the summer fire while the six *Hakea sericea* individuals were killed. Indigenous species were present in all treatments including the Control (Fig. 1a). While basal diameters of indigenous plants did not vary significantly between treatments, a number of basal diameter outliers were present within the Fynbos, Mix and Terraces treatments but mainly absent in the Control plots (Fig. 1a). Invasive species were generally larger and occurred at higher densities than indigenous species on the restoration site (Fig. 1b) with a relatively-equal size distribution of basal diameters across the treatments.

A few woody species dominated the Oaklands site prior to the 2005 fire (Fig. 2), viz. *A. mearnsii*, *B. lanuginosa*, *P. laurifolia* and *P. repens*. Fewer indigenous woody individuals occurred within the Control where no seeds were sown while the number of *A. mearnsii* did not differ significantly between treatments ($F=0.36$, $P=0.78$, Kruskal–Wallis: $P=0.86$). The Fynbos treatment hosted a substantially larger number of dominant woody fynbos individuals (458) compared to the other two sowing treatments (Mix: 210, Terraces: 214).

The 2006 size-class distribution of *A. mearnsii* showed the typical pattern created by continuous recruitment (Fig. 3a). While *P. laurifolia* (Fig. 3b), *P. repens* (Fig. 3c) and *B. lanuginosa* (Fig. 3d) are resilient to fire events and should show event-driven seedling recruitment triggered by fire disturbance, all of these adults were dead, but no seedlings were recorded in 2006 (refer to Section 3.2). These figures confirm the absence of indigenous woody individuals within Control plots and also indicate that Fynbos treatment plots had the largest number of indigenous individuals. Indigenous species occurred sparsely (if at all) within the Control plots.

Table 2 shows the basal diameter means and standard deviations of all the woody species recorded and identified within the different treatments in 2006 (whether dead or resprouting). Where only one specimen was recorded for some indigenous species listed, no data are shown. All the invasive tree species

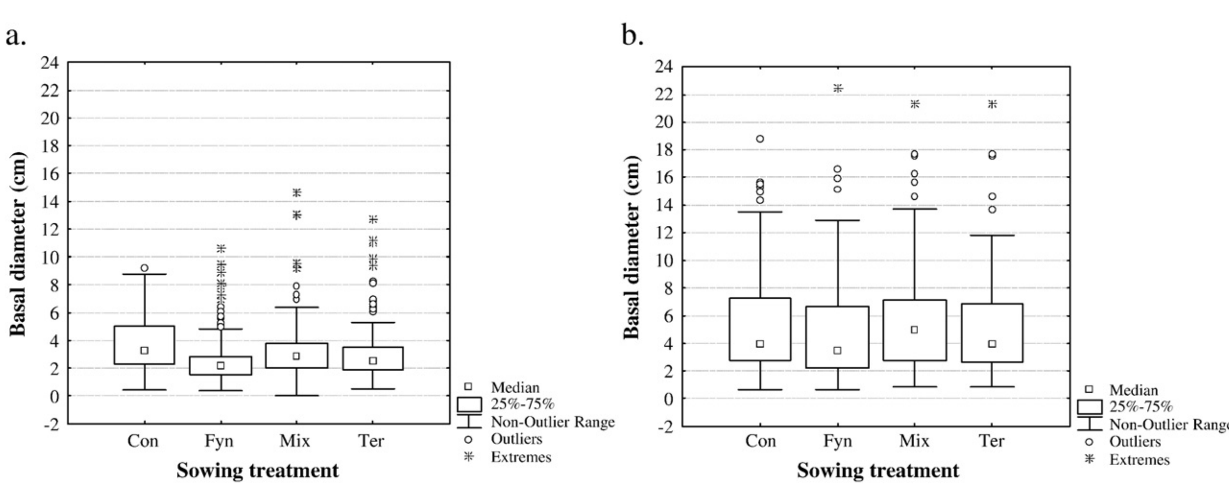

Fig. 1. Basal diameters of (a) indigenous and (b) invasive woody species recorded in 2006 within each sowing treatment respectively. Con = Control, Fyn = Fynbos treatment, Mix = Mix treatment, Ter = Terraces treatment.

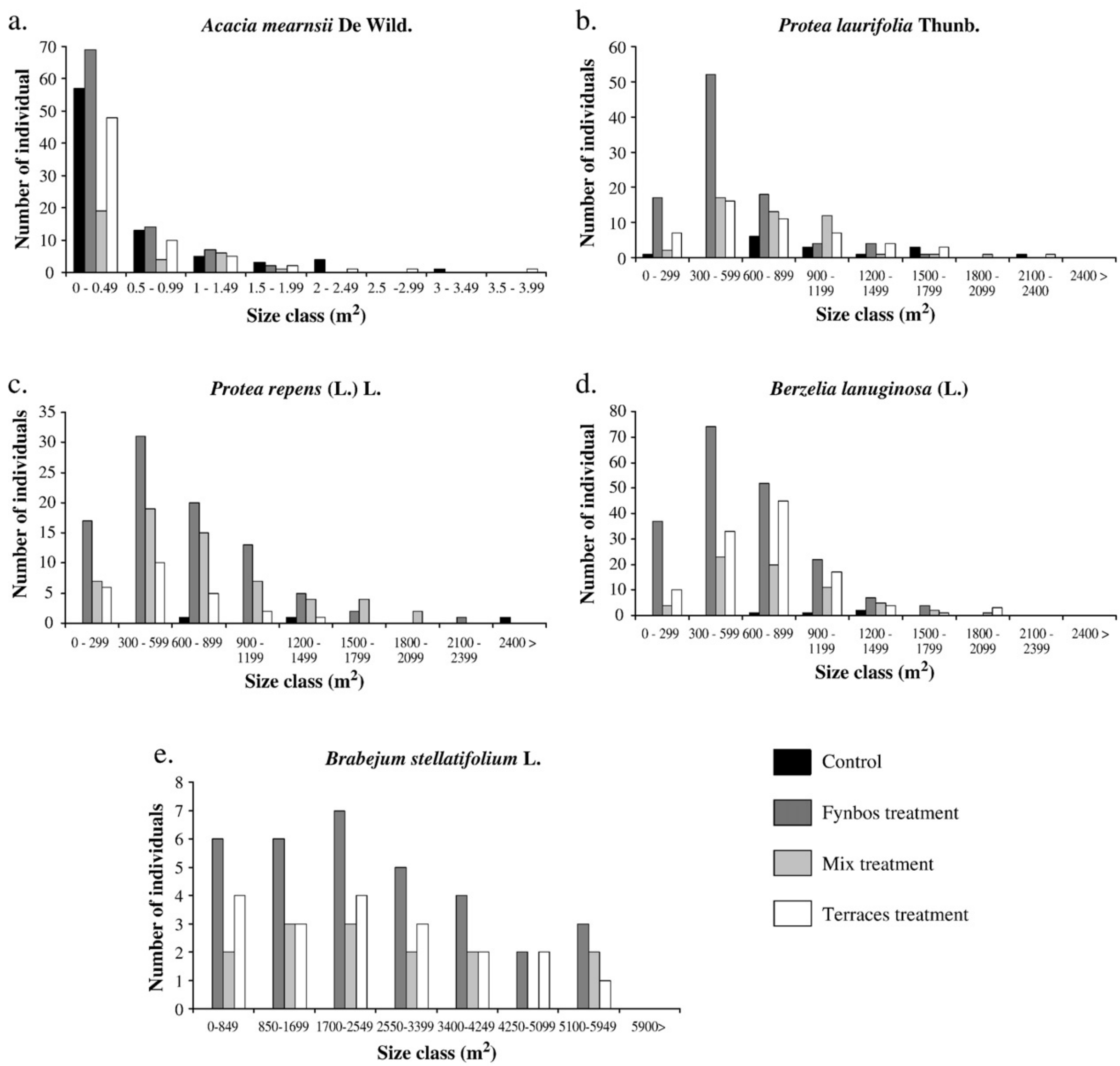

Fig. 3. Size-class distributions (height, m) of (a) *Acacia mearnsii*, (b) *Protea laurifolia*, (c) *Protea repens*, (d) *Berzelia lanuginosa* and (e) *Brabejum stellatifolium* adults recorded in the 2006 survey. Individuals were either dead or resprouting, however size classes (basal diameter × heights) were indicative of pre-fire population status.

present on the restoration site were represented in the Control plots. Only *A. mearnsii* occurred within all treatments while *E. cladocaylyx* and *H. sericea* occurred in two other treatments respectively. *B. lanuginosa* had a much larger mean basal diameter within the Control plots than within other treatments. However, it must be noted that there were only 4 individuals in the Control plots (too few to be relevant in a statistical analysis) while the other treatment plots had over 50 individuals (see Fig. 3d). The Fynbos treatment had the smallest basal diameters of *B. lanuginosa* but also had the largest number of individuals (196) while the means within the Mix and Terraces plots were almost the same with no significant difference between them (Bonferroni test, $P = 1.00$). A difference between the Fynbos and the Terraces treatments was detected using the Bonferroni test but it was not statistically significant ($P = 0.138$). *P. laurifolia* had the largest basal diameters within the Control plots, but while a difference was detected (via a Bonferroni test) between the Control and Fynbos treatments, it was not statistically significant ($P = 0.085$). Again the lowest number of plants occurred within the Controls (15). No significant differences were found between the basal diameter means of any other species recorded.

3.2. Seedling survey (including Bracken)

Rank-abundance curves of the seedling survey data in 2006 (Fig. 4) showed a strong dominance of a few species with large numbers of individuals within the 1 m^2 survey plots. A total of 47 species were recorded and identified. *A. aethiopicum*, used as mulch and as a seed source in the experiment, was the most abundant species within the three sowing treatments. *Gnidia tomentosa*, an indigenous perennial shrub that originated from

Table 2
Basal diameter means and standard deviations of the woody species within the different treatment (50 m^2) plots in 2006

Species Name	Control treatment	Fynbos treatment	Mix treatment	Terraces treatment
Woody invasive alien species				
Acacia mearnsii De Wild.	6.89±3.02	6.37±4.25	7.40±3.65	5.66±1.39
Pinus spp.	3.87±3.16	–	–	–
Eucalyptus spp.	5.99±5.89	9.31±5.06	–	–
Hakea sericea Schrad. & J.C. Wendl.	2.09±1.81	–	–	2.71±1.13
Woody indigenous species				
Berzelia lanuginosa (L.) Brongn.	3.97±1.48	2.00±0.26 [a]	2.38±0.38	2.40±0.16 [a]
Brabejum stellatifolium L.	–	6.04±2.89	7.80±4.02	6.62±1.59
Diospyros glabra (L.) De Winter	1.62±1.77	1.35±0.74	–	3.50±0.00
Leucadendron salicifolium (Salisb.) I. Williams	–	5.41±2.21	6.10±2.10	6.83±1.04
Metrosideros angustifolia (L.) J.E. Sm.	5.73±1.72	–	4.27±1.14	–
Myrsine africana L.	–	0.60±0.19	–	–
Protea laurifolia Thunb.	4.41±1.42 [a]	2.84±0.16 [a]	3.04±0.94	3.18±0.58
Protea repens (L.) L.	4.29±1.25	3.01±0.43	3.25±0.72	2.98±1.11
Rhus angustifolia L.	2.69±0.71	1.91±0.85	–	2.86±0.90
Rhus rehmanniana Engl. var. *glabrata* (Sond.) Moffett	2.03±0.56	1.84±0.68	–	1.93±0.65

[a] Indicates where differences were detected between treatments for a particular species but none were statistically significant.

the seed bank, was the second most abundant species within all sowing treatments.

Table 3 indicates the full data set for plant (seedlings, but including Bracken) densities in each treatment over time while Table 4 provides a statistical comparison (where possible) of densities over time (combining all treatments) and across treatments (combining time). *P. aquilinum* subsp. *aquilinum* (Bracken) was the dominant ground cover under the acacia canopy prior to the clearing treatment and also was the first to emerge after the clearing treatment fire in 1998 (Prins, 2003). Mean density comparisons over time revealed a significant difference in Bracken density between 1998, 1999 and 2006 ($F=16$, $P<0.001$) with the highest density recorded in 1998, the lowest density recorded in 1999 (Table 4). Although no statistically significant differences were found between the treatments over time (Table 4), it is useful to mention that a large number of *A. mearnsii* seedlings emerged among the other seedlings in 2006 with the most emerging in the Control plots (Table 3) and the least within the Fynbos plots. When the 1998 and 2006 data were statistically compared, the number of acacia seedlings increased significantly in the Control treatment while densities stayed more or less the same with no significant differences within the Fynbos, Mix and Terraces treatments over time.

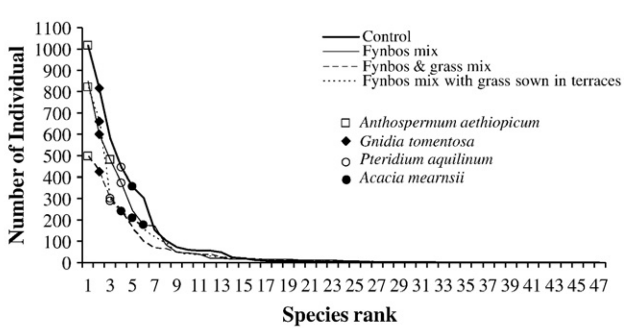

Fig. 4. Rank-abundance curve of seedlings within 1 m^2 plots in Control, Fynbos, Mix and Terraces treatments in 2006.

Ericoid density differed significantly over time ($F=10.49$, $P<0.001$) (Table 4) with significant differences between 1998 and 1999 ($P<0.001$), and 1999 and 2006 ($P=0.036$). No Proteaceae seedlings were recorded in 2006 (Tables 3 and 4), which gave rise to the significant differences in mean seedling densities between 2006 and 1998 and 1999 respectively ($P<0.001$). There was also a significantly-lower density of Proteaceae seedlings in the Control compared to the other three treatments over time (Table 4). Indigenous grass densities showed no significant change between the treatments overtime. However, alien grass densities (including sown and other species) changed significantly over time and differed significantly between treatments (Table 4). The two alien grass species that were used in the sowing treatments in 1998 were not recorded in 2006, however, other alien grass species were recorded. Forb density remained stable except for a significant increase in alien forbs from 1998 and 1999 to 2006 (Table 4). Broad-leaved shrubs also increased significantly from 1998 and 1999 to 2006 and a significant difference occurred between the Control and Fynbos treatments (Table 4). Geophyte density increased significantly from 1999 to 2006 and differed significantly between the Fynbos and Terraces treatments. The total covers recorded in all three years differed significantly from each other, where 1999 had the highest, 2006 the lowest and 1998 an intermediate mean total cover. No significant differences for total cover were detected between treatments. Indigenous cover increased significantly from 1998 to 1999; however, no estimate could be made for alien cover because of a lack of sufficient data for all treatments.

4. Discussion

Eight years after initial clearing, *A. mearnsii* still had a strong presence and remained one of the dominant species at the restoration site. This is because the site received only one

Table 3
Mean plant densities (±standard deviations) and % cover in 4 quadrats (1 m^2) for Control and sowing treatments (Fynbos, Mix, Terraces) in 1998, 1999 and 2006

Growth form guild	Control			Fynbos treatment			Mix treatment			Terraces treatment		
	1998	1999	2006	1998	1999	2006	1998	1999	2006	1998	1999	2006
Ericoids	42.6± 52.47	7.3± 1.64	50.95± 47.24	75.5± 77.48	15.08± 6.72	25.05± 33.99	54.30± 45.45	11.86± 4.11	27.05± 23.91	46.10± 35.96	12.71± 4.62	41.60± 55.34
Proteoids	0.05± 0.23	0.7± 1.30	0.00± 0.00	7.40± 5.84	6.36± 3.36	0.00± 0.00	4.90± 4.00	4± 1.49	0.00± 0.00	6.46± 6.06	4.2± 2.31	0.00± 0.00
Acacia mearnsii	7.65± 7.24	–	17.90± 18.99	9.11± 10.44	–	9.03± 4.51	9.86± 8.00	–	11.95± 10.69	10.50± 13.32	–	10.50± 4.57
Eucalyptus supp.	4.43± 2.23	–	0.10± 0.14	1.35± 0.63	–	0.70± 1.43	0.95± 1.32	–	0.00± 0.00	1.20± 1.24	–	0.00± 0.00
Indigenous grasses	2.35± 3.20	22.76± 24.41	41.95± 45.83	5.55± 10.66	23± 26.20	46.60± 67.49	40.50± 33.88	5.5± 10.95	9.60± 8.22	56.00± 36.51	9.26± 9.06	13.05± 11.63
Alien grasses	0.15± 0.67	1± 2.24	0.40± 0.29	0.00± 0.00	0.75± 1.12	0.25± 0.25	57.50± 61.63	97.9± 49.20	0.05± 0.11	17.00± 45.20	51.20± 24.34	0.10± 0.22
Indigenous forbs	2.35± 3.13	4.16± 7.91	3.55± 3.63	2.53± 3.34	3.05± 2.29	3.05± 2.52	1.80± 2.21	1.25± 1.72	4.60± 4.79	2.00± 3.21	1.15± 0.96	9.45± 7.98
Alien forbs	1.00± 1.45	2± 0.77	21.80± 31.35	0.80± 1.28	2.15± 1.36	7.70± 6.76	3.25± 10.69	1.05± 0.93	6.05± 2.75	0.65± 1.42	1.9± 1.46	10.40± 9.05
Broadleaved shrubs	0.00± 0.00	0.2± 0.33	1.25± 3.65	0.05± 0.22	0.05± 0.11	19.35± 36.06	0.00± 0.00	0.00± 0.00	0.15± 27.64	0.45± 1.61	0.05± 0.11	0.00± 28.6
Bracken	29.1± 18.63	10.82± 9.18	22.3± 10.69	37.0± 25.17	9.77± 4.31	18.70± 10.33	20.20± 20.25	8.01± 7.24	15.10± 12.14	22.10± 23.7	8.81± 8.54	14.35± 8.18
Geophytes	0.40± 0.75	2± 3.93	2.3± 2.27	0.70± 1.42	0.5± 0.87	15.45± 17.54	0.20± 0.52	0.05± 0.11	3.90± 5.04	0.30± 0.92	0.1± 0.22	2.40± 3.18
Alien cover	9%	9%	–	8.20%	–	–	19%	20%	–	11%	17%	–
Indigenous cover	30%	51%	–	39%	51%	–	23%	40%	–	24%	41%	–
Total cover	35%	51%	30%	42%	52%	30%	37%	52%	17%	31%	51%	24%

Statistical comparisons are given in Table 4.
Ericoids include *Anthospermum aethiopicum* that was introduced to all plots in the mulch material.

follow-up clearance in early 1999 and individuals continued to recruit from the soil seed bank. However, by comparing the Control and sown plots, a substantial difference in the presence and abundance of indigenous fynbos species was noted, with greater density and diversity of indigenous species in the sown plots. The small number of indigenous woody plants (an average of 11.8 plants per 50 m^2) within the Control plots could be the result of seed movement from adjacent sown plots (most likely in the case of serotinous species) or recruitment from a seed bank.

The smaller number of woody plants in the Control plots, and therefore potentially lower levels of competition, may be why woody indigenous plants were generally larger in the Controls compared to the other sowing treatments. In contrast,

Table 4
Statistical comparison of mean plant densities (±standard deviations) and % cover (N=4, 1 m^2 quadrats) over time, for all treatments combined (1998, 1999 and 2006) and across treatments (Control, Fynbos, Mix, Terraces) with years combined

Growth form guild	Years			Treatments			
	1998	1999	2006	Control	Fynbos	Mix	Terraces
Ericoids	54.64±6.71^{a}	11.74±1.04^{b}	36.17±9.37^{a}	–	–	–	–
Proteoids	4.70±0.51^{a}	3.82±0.51^{a}	0.00±0.00^{b}	0.25±0.61^{y}	4.58±0.61^{x}	2.96±0.61^{x}	3.55±0.61^{x}
Acacia mearnsii	–	Not recorded	–	–	–	–	–
Eucalyptus spp.	–	Not recorded	–	–	–	–	–
Indigenous grasses	–	–	–	–	–	–	–
Alien grasses	18.65±4.9^{a}	37.79±6.14^{b}	0.20±0.05^{c}	0.52±5.79^{x}	0.33±5.79^{x}	51.8±5.79^{y}	22.87±5.79^{x}
Indigenous forbs	–	–	–	–	–	–	–
Alien forbs	1.43±0.60^{a}	1.78±0.26^{a}	11.53±3.73^{b}	–	–	–	–
Broadleaved shrubs	0.13±0.11^{a}	0.08±0.04^{a}	34.82±6.01^{b}	1.75±3.99^{y}	20.19±3.99^{x}	10.30±3.99^{x}	14.45±3.99^{x}
Bracken	27.07±4.41^{a}	9.35±1.69^{b}	17.62±2.33^{c}	–	–	–	–
Geophytes	1.64±1.22^{a}	0.66±0.45^{a}	6.03±2.09^{b}	1.57±1.57^{x}	7.20±1.57^{y}	1.38±1.57^{x}	0.95±1.57^{x}
Alien cover	–	–	Not recorded	–	–	–	–
Indigenous cover	29.25±2.90^{a}	45.75±2.15^{b}	Not recorded	–	–	–	–
Total cover	36.25±2.84^{a}	51.05±1.64^{b}	24.90±2.57^{c}	–	–	–	–

Letters in superscript show where significant differences exist (Repeated Measures ANOVA: P<0.05, Tukey HSD test: P<0.05). Values with different letter superscripts are significantly different from one or more of the other values. Significant differences are indicated as abc between Years and xyz between Treatments. Where there are no data inserted, no significant differences occurred.

basal diameters of the woody invasive trees were similar across all sowing treatments, indicating that these species are less affected by competitive interactions. *A. mearnsii*, among other invasive trees, possesses competitive abilities to enhance nutrient acquisition and to obtain scarce resources through mechanisms of nitrogen fixing symbionts, sheathing mycorrhizas and extensive root production (Stock and Allsopp, 1992).

A census of seedlings over time showed that restoring the site after alien clearance, by sowing indigenous seeds, increased both diversity (by improving species presence) and abundance. However, the 2006 results also showed that a few sown species dominated the restoration site. While indigenous diversity was not fully restored, this provides a good indication of what species should be sown to facilitate vegetation recovery. A seed mix including only fynbos and riparian species appears to improve diversity more efficiently than a mix including alien annual grass. Nevertheless, all three sowing treatments proved to suppress *A. mearnsii* seedling recruitment over time. Thus revegetation of sites after invasive alien clearance may indeed help to minimize re-invasion (Holmes and Richardson, 1999; Holmes et al., 2005).

The actively-restored community appeared to be resilient to fire. Three fynbos species showed event-driven recruitment while most species with resprouting capability resprouted after the December 2005 fire. In addition, total seedling density of the community increased over time, which is also an indication of the community's recovery progress and resilience to fire. However, *A. mearnsii* still co-dominated the site as it is also resilient to fire disturbance and shows continuous recruitment.

The absence of Proteaceae seedlings after the December 2005 fire was unexpected as many of the skeletons of adults bore cones, indicating that a canopy held seed bank (serotiny) may have been present. This recruitment failure may be due to several reasons including: 1) low viable seed production (*Proteas* were smaller than average owing to competition with taller woody invasive species); 2) seed mortality during the fire (as the invasive canopy is taller and the cones were no longer held above the higher fire temperatures, as would occur in uninvaded veld); 3) pollination failure resulting in inflorescences but no seeds (lack of visibility among the invading plants) or 4) seeds were eaten by seed predators (because there were too few to satiate seed predators).

Reseeding species are generally killed by fire disturbances and regenerate only from the seeds released from the canopy after fire (Bond et al., 1984, 1995). Seeds are unlikely to remain viable for longer than a year on the soil surface.

Sowing a mixture of indigenous seed and including alien grass seed appears to inhibit *A. mearnsii* seedling recruitment to some extent, although not entirely. Since this was a small-scale restoration experiment and the stream sub-catchment was still invaded by acacias, *Acacia* soil seed banks were not exhausted during the 1998 fire. *A. mearnsii* typically has a large, persistent seed bank that accumulates because of the prolific numbers of seed set per growing season. The seeds are long-lived unless stimulated by a disturbance, especially fire (Pieterse and Boucher, 1997). However in the sown treatments, establishment and growth of *Acacia* after the 1998 fire may have been suppressed, resulting in lower *Acacia* seed production and therefore lower seedling recruitment after the 2006 fire.

Introducing seed mixes also seems to significantly reduce the number of *E. cladocaylyx* and *P. aquilinum* subsp. *aquilinum* recruits. The decrease in density of Bracken means that other seedlings have a chance to recruit because the dominance of *P. aquilinum* subsp. *aquilinum* could suppress the regeneration of other fynbos plants (Prins, 2003). This is an important observation since several native species, if favoured by management and/or invasion history, have the potential to radically change conditions and to alter the outcome of restoration.

It is interesting that the indigenous grass density was significantly reduced in the plots where the alien grasses were sown, while increased in the Control and Fynbos treatments. Thus, there is a clear dynamic interaction between alien and indigenous grasses within the different treatments. It also appears that the density of the alien grasses that were sown did not increase to a threatening state, as may have been predicted from the 1998 consecutive survey data. Alien grass density was very low in the 2006 census, and non-existent for the two annual species used in the sowing treatments, indicating a lack of medium-term persistence of these species. This confirms that they may safely be used for short-term soil stabilization where indigenous seed is scarce.

Thus, returning to our original aims, we have shown that by sowing a mixture of seeds of indigenous plant species, the recovery of riparian vegetation density and cover is possible. Furthermore, we demonstrate that re-established indigenous vegetation cover can reduce the recruitment of woody invasive *A. mearnsii* seedlings. While the sowing of non-invasive alien grasses may have assisted recovery by suppressing woody invasive seedlings, we noted a dynamic interaction between alien and indigenous grasses within the different treatments. The treatment most effective in promoting the recruitment of indigenous species was that of a sowing treatment comprised of evenly-sown seeds of local fynbos and riparian species. The restoration treatments were resilient to fire; however, without adequate alien follow-up control, re-invasion from seed banks will remain a threat to any restoration initiative.

This is the first report of a riparian restoration trial using active species re-introduction in the Fynbos Biome. However, the trial was done on a small scale and focused entirely on ecological implications. Further research integrating ecological, economic and social aspects of ecosystem repair after alien clearing is clearly needed.

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