Re-evaluation of a riparian restoration experiment in the Western Cape Province: status 8 years down the line

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Abstract

In 1998, a study was initiated to assess the relative effectiveness of three different sowing treatments for reducing soil erosion and restoring indigenous riparian vegetation cover after alien clearing in the Western Cape. The study concluded that introducing a mixture of seed to a riparian zone with a history of invasion can increase indigenous vegetation cover and species richness and has a stabilizing effect on eroded riverbanks. We report on a re-assessment of the restoration plots after a period of 8 years (with no follow up control) and a summer fire in 2005. The re-assessment included a survey of all the burnt woody material within the different treatment plots to determine species presence (dead or resprouting) and abundance, as well as a survey of seedlings in a post-fire environment. Woody invasive plants dominated all plots, and had survived the burn with the majority re-sprouting, indicating the dire need for follow-up control to justify initial clearing and restoration cost. However the re-assessment revealed that indigenous vegetation cover was considerably higher in the plots that had received sowing treatments, while the control plots were dominated by *Acacia mearnsii*, the main invader of the tributary. Restoration of riparian areas after alien plant clearing has potential, but must be coupled with a long-term plan for follow up removal of post clearing alien recruits.

Keywords: Alien clearing, Riparian vegetation, Restoration, Follow-up treatment

Introduction

The invasion of natural ecosystems by invasive alien plant species has detrimental effects on agriculture, forestry and human health (Walker & Steffen, 1999; Wilcove et al., 1998). It is also recognized as the second-largest global threat to biodiversity (after direct habitat destruction) (Walker & Steffen, 1999; Wilcove et al., 1998; Rebelo, 1992). River systems in South Africa have been extensively invaded by invasive alien plants with river banks and river beds being 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). Habitat disturbances have been found to promote invasion by exotic plant species (Hobbs, 1989; Mack, 1989; Ewel, 1986; D'Antonio et al., 2002; Holmes, in prep). Invasive alien plants may be introduced to riparian zones by human mediated disturbance while the spreading of propagules within catchments may be facilitated by the movement of water and bare soil patches created by natural disturbances (Hood and Naiman, 2000). Thus, riverine ecosystems are naturally more susceptible to invasion by alien species than other ecosystems (Hood and Naiman, 2000).

Riparian zones are relatively narrow, linear features in the landscape, except for broad floodplain systems and form a transition zone between aquatic and terrestrial ecosystems (Gregory et al., 1991). The vegetation within riparian zones differs in structure and function from vegetation types within other ecosystems (Naiman and Décamps, 1997; Prins et al., 2004). Riparian vegetation has important functions within the landscape (Malason, 1993; Naiman and Décamps, 1990, 1997) in that it controls the flow of water, nutrients and sediments between the aquatic and terrestrial ecosystems (Décamps, 1993; Junk et al., 1989; Forman and Godron, 1986; Peterjohn and Correll, 1984), it serves as a landscape corridor (Baker et al., 1993; DeFerrari and Naiman, 1994; Forman and Godron, 1986; Pysek and Prach, 1994; Samways, 2005) but also provides habitat (Stewardson et al., 2004) and act as a barrier or filter for different species (Samways, 2005). Because it is a dynamic ecosystem with changing hydrological cycles, riparian zones have disproportionately high species richness to other landscape forms (Murray and Stauffer, 1995; Naiman et al., 1993). Riparian vegetation has the vital role of determining the structure and stability of the stream banks and thereby preventing erosion. Riparian vegetation also plays a role in that it may shade the aquatic ecosystem and thereby have an influence on water temperature and primary production within streams (Stewardson et al., 2004).

In cases where riparian zones are invaded by invasive alien plant species, mostly woody tree species in the Western Cape, the diverse riparian vegetation are replaced with species poor alien stands (Cronk and Fuller, 1995; Richardson et al., 1989 & 1997; Richardson and van Wilgen, 1986 & 2004; Cowling et al., 1976) by reducing local biodiversity (Richardson et al., 1992). Ecosystem functions like nutrient and hydrological cycling, geomorphological processes (Cronk and Fuller, 1995; Richardson et al., 1997; Richardson and van Wilgen, 2004) and physical recourses (Vitousek, 1990) are altered to a great extent by alien invasions. Alien plant invasions significantly reduce indigenous seed-bank density and richness and cause changes in both seed-bank composition and guild structure (Holmes, 2001; Macdonald and Richardson, 1986). Dynamic interactions exist between invasive alien plants and indigenous plant species of which seed-bank dynamics are greatly affected and influenced by fire and clearing treatments (Macdonald, 2004). The degree to which a riparian ecosystem recovers naturally to an indigenous state after alien clearing are determined by the vegetation and soil types, the alien plant species involved and the infestation age and density, the number of fire cycles experienced by the infestation prior to and after clearing and the clearing treatments used (Macdonald, 2004). Riparian vegetation also recovers slower after a dense infestation has been cleared in comparison to the recovery of a lightly infested area (Richardson et al., 1989). Invasive alien species builds up seed banks which complicate long-term restoration of a site and challenges management goals (D'Antonio et al., 2002). However, managers do use exotic species to restore particular functions within a degraded landscape in circumstances where native species cannot deliver the desired function (D'Antonio et al., 2002). In the Oaklands restoration study in 1998, Prins used alien grass species to grow quickly and stabilize the soil banks of the river (Prins, 2003)

A major concern is that some invasive alien trees are believed to consume substantially more water than the native vegetation (Dye and Jarmain, 2004; Görgens and van Wilgen, 2004). Since South African water resources are scarce, the South African government instigated the Working for Water program in 1995 (Richardson and van Wilgen, 2004) to clear the watersheds of invasive trees to enhance stream flow (Dye and Jarmain, 2004; Dye et al., 2001). Labour intensive approaches are taken with dense infestations (Vosse, in prep) (as was the case of the Oaklands site in 1998 (Prins, 2003)). The stands are felled using mechanical and chemical alien control methods and left for 12 – 18 months to allow for seed release and germination, followed by burning of the site to kill off all alien seedling recruits (Vosse, in prep). The site should then be visited regularly at 1.5 – 2 year intervals to remove all invasive seedlings present (Vosse, in prep). This method proves to be very successful (Holmes, 1998). However a limited budged prevents the Working for Water initiative from fully restoring the indigenous vegetation through restoration projects and this often leads to unsatisfactory and inadequate results after the clearing of an infested area (Holmes, 1998; Holmes, 2001). It is important to restore native vegetation as quick 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).

The aim of ecological restoration is to restore diversity and ecosystem dynamics caused by human-induced changes to an ecosystem (Jackson et al., 1995). 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). Since riparian habitats are dynamic ecosystems moulded by hydrological fluctuations (Planty-Tabacchi et al., 1996; Shafroth et al., 2002), the main target of riparian restoration must be to restore the rivers hydrological regime (flood frequency and intensity) (Ehrenfeld, 2000; Vaselaar, 1997; Patten et al., 2001; Rood et al., 2003). The Working for Water initiative plays a critical role in ecosystem rehabilitation and restoration by removing thirsty aliens from riparian zones to enhance stream flow. Riparian species are adapted to colonizes appropriate sites created by floods and sediment movement (Wissmar and Beschta, 1998). 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 could be the target for ecosystem repair (Hobbs and Harris, 2001).

Many riparian specialist species are relatively widespread and 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 insures 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 at the Oaklands site. It has been suggested by Prins et al. 2004 that emphasis should be placed on reintroducing 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. This has been the case of the 1998 Oaklands riparian restoration study as mostly generalist species have been used.

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 (Holmes et al., 2005). This is because little monitoring has been done (Holmes et al., 2005). This motivated the re-assessment survey at Oaklands after 8 years since the restoration effort in 1998. The objective of the 1998 study was to test the effectiveness of three different sowing treatments to restore indigenous vegetation cover and reduce soil erosion in the effort to rehabilitate the riparian zone after alien clearing. The re-assessment survey concentrates on the success rate to which indigenous vegetation cover has been restored and the degree to which alien re-colonization has been suppressed by re-vegetating the site. We hoped to find a reduction in the number of acacias present on the site with the indigenous riparian community well established and resilient to fire regimes.

2. Materials and methods

2.1 The study area

The study site is situated on the farm Oaklands at the foot of Groenberg near Wellington in the Western Cape (33˚36’80’’S/19˚5’30’’E). The vegetation type of the catchment is mountain fynbos growing in soils derived from deeply weathered granites with the upper layers containing colluvial sandstone material (Le Maitre et al., 1996). A small, alien invaded stream that was subject to some CSIR studies in the past (Dye et al., 2001; Dye and Jarmain, 2004), was chosen for the study. *Acacia mearnsii* De Wild. and *Eucalyptus grandis* W.Jill ex Maiden invaded the whole catchment area while *Acacia mearnsii* De Wild. dominated the riparian zones (Prins, 2003). Working for Water teams clearfelled the riparian zones by felling the alien trees without removing the tree boles or slashed material (Prins, 2003). A wild fire swept through the clearfelled area on Oaklands farm early in 1998 (Prins, 2003). This gave rise to a clearing treatment of fell and burn within which the 1998 experiment was set up (Prins, 2003). No follow-up clearing treatments were applied and the alien trees were left to re-colonize the catchment. A natural summer fire occurred in the catchment during 2005/’06. A visit to the site in April 2006 revealed that various mature indigenous and invasive woody specimens were present within the study area.

2.2 Experimental design

The experimental layout of the study done in 1998 consisted of twenty continuous, 50 m^2 plots set up on the north facing bank within the riparian zone of the selected stream. Each 50 m^2 plot received one of three different sowing treatments (hereafter named the Fynbos, Mix and Terraces treatments) or a Control treatment. The Fynbos sowing treatment comprised of evenly sown seeds of chosen fynbos and riparian species. The Mix sowing treatment comprised of evenly sown seeds of the same fynbos and riparian species including two grass species. The Terraces treatment included all fynbos, riparian and grass species used but the way of sowing the seeds differed in that the fynbos and riparian seed mix were evenly sown while the grass seeds were sown in terraces. The treatments were randomly allocated to the plots with five replicates each. The chosen fynbos and riparian species included within the seed mix were the fynbos species; *Euryops abrotanifolius* (L.) DC., *Rhus angustifolia* L., *Diospiros glabra* (L.) De Winter, *Protea repens* (L.) L., *Protea laurifolia* Thunb., *Leucadendron salignum* P.J.Bergius., *Pentaschistis curvifolia* (Schrad.) Stapf, *Athanasia* spp., *Stoebe* spp., *Tribolium* spp., *Montinia* spp., *Helichrysum* spp., *Anthospermum aethiopicum* L., and riparian species; *Berzelia lanuginosa* (L.) Brongn., *Leucadendron salicifolium* (Salisb.) I.Williams and *Brabejum stellatifolium* L.. The annual grass species used in the experiment were *Eragrostis tef* (Zucc.) Trotter and *Avena fatua* L.. *Anthospermum aethiopicum* L. was used as mulch as well as seed source and all plots were covered with it immediately after the treatments were applied 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.

2.3 Data collection

The data collection done in 1998 consisted of five separate censuses 5, 6, 9, 13, and 15 months after sowing. For the purpose of this re-assessment, we used the data of the first census, five months after sowing, which was carried out on the 1st, 2nd and 8th of October 1998 after the first rainy season since the treatments were applied. This particular data was used to try and match the season the data was collected in 2006. The re-assessment census took place from the 11th to the 14th of September 2006 to ensure significant seedling recruitment after the winter rains. All plots were still marked with the original pegs used in the 1998 study.

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 for dead or re-sprouting and their heights estimated. Diameter measurements were also taken at the base and breast height (1.3 m above ground).

To enable comparisons with the 1998 study, the methods used by Prins (2003) were replicated in the 2006 re-assessment survey of recruiting seedlings. All seedlings were identified and classified into 11 growth form guilds: ericoids, proteoids, acacias, eucalyptus, alien forbs, indigenous forbs, alien grasses, indigenous grasses, broad leaved shrubs, bracken and geophytes. Density counts were made for each guild and the total cover was estimated for each 1 m^2 plot. A new growth form guild namely "indigenous shrubs" was created for a couple of species that could not be assigned to a guild identified in the 1998 study.

Another aim was to do a seedling to parent ratio with *Protea repens* (L.) L. and *Protea laurifolia* Thunb. but unfortunately no protea seedlings could be identified. It may be that the seedlings are still dormant and are still to emerge.

2.4 Data analysis

The woody material recorded within the 50 m^2 plots was analysed using various analytical methods. Box plots were constructed using the basal diameter data of the indigenous and invasive woody species respectively with the aim to compare plant growth between treatments. A rank-abundance curve was constructed using all woody species recorded to describe evenness of species distribution and relative species dominance within the different communities of each sowing treatment. Size-class distributions were constructed for *Acacia mearnsii* De Wild, *Protea repens* (L.) L., *Protea laurifolia* Thunb., *Berzelia lanuginose* (L.) Brongn. and *Brabejum stellatifolium* L. for each sowing treatment respectively. This was done by multiplying the basal diameters (cm) with the estimated heights (converted to cm). Equally sized intervals were used to group the results. The size-class distributions enable detection of recruitment patterns of the species and comparison of the number of individuals within each sowing treatment. Single factor ANOVA tests were done to see if there was any significant difference ($p<0.05$) in basal diameter for each species recorded between the sowing treatments. T-tests (two-sample assuming unequal variance, two tail) was used to determine which treatments differed significantly ($p<0.05$) from each other. The aim is to find size differences of a species between treatments to examine the competition within and between species, the availability of nutrients and hence the number of individuals present competing for nutrients.

The data of the seedling survey within the small 1 m^2 plots were used to construct a table showing the means and standard deviations of the density recorded for each growth form within the different sowing treatments. The 2006 mean densities are then compared to the data Prins recorded in 1998 using T-tests (two tail, $p<0.05$) to see if there are any changes of densities and/or growth form dominance within the plots. A rank-abundance curve was used to determine the evenness of the species distribution and relative species dominance.

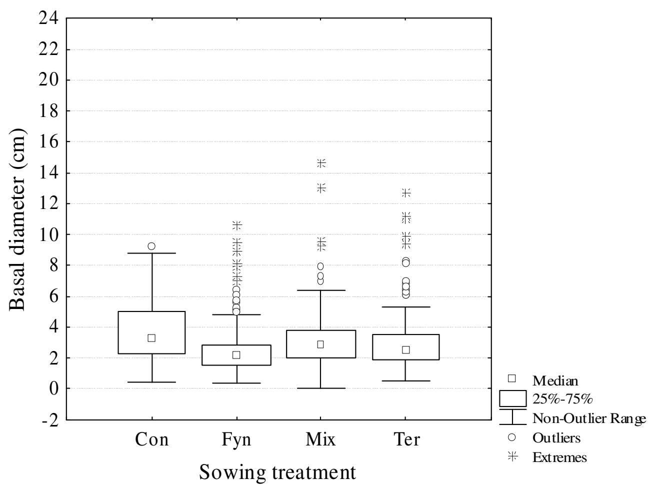
Results

Most of the species used for the sowing treatments in 1998 were identified on the site during the re-assessment survey except for four fynbos species and the two alien grass species used to stabilize the soil and reduce soil erosion. A lot of species were recorded both in 1998 and during the re-assessment survey that were not sown but emerged from the seed bank.

Woody material

During the re-assessment survey it was noted that *Protea repens* (L.)L. and *Protea laurifolia* Thunb. adult plants were killed by the 2005 summer fire. All recorded individuals of the species *Olea capensis* L. subsp. *capensis*, *Leucadendron salignum* P.J.Bergius, *Rhus angustifolia* L., *Rhus rehmanniana* Engl. var. *glabrata* (Sond.) Moffet, *Myrsine africana* L., *Podalyria calyptrata* (Retz.)Willd., *Olea europaea* subsp. *africana* , *Metrosideros angustifolia* (L.) J.E. Sm. and *Maytenus oleoides* (Lam.) Loes. were resprouting. More than 90 percent of the *Brabejum stellatifolium* L. and *Diospiros glabra* (L.) De Winter individuals recorded were resprouting. All *Acacia mearnsii* De Wild. and *Eucalyptus grandis* W.Jill ex Maiden individuals were still alive after the summer fire while the six *Hakea sericea* Schrad. & J.C. Wendl. individuals were killed.

a.



b.

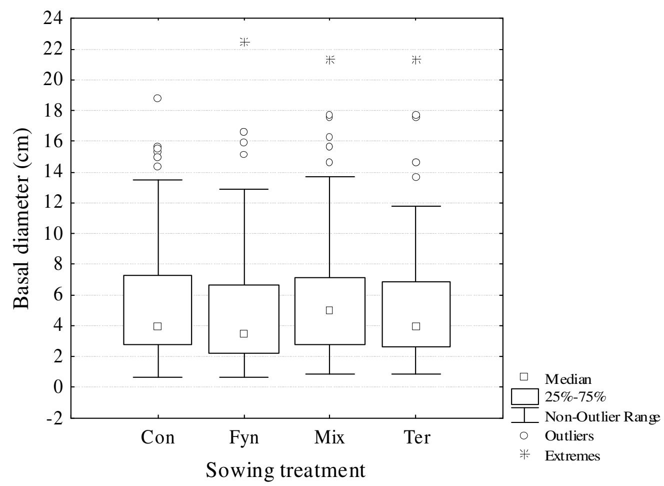


Fig. 1. Box plot diagrams of the basal diameters of the indigenous (a) and invasive (b) woody species within each sowing treatment respectively. Con = Control treatment, Fyn = Fynbos treatment, Mix = Mix treatment, Ter = Terraces treatment.

The box plots in fig. 1a show that indigenous species are present within the control plots. The basal diameters of the indigenous plants within the control plots are also somewhat larger compared to the other treatments However, there are a number of outliers present within the fynbos, mix and terraces treatment plots while outliers are mainly absent in the control plots. Fig. 1b shows a relatively equal size distribution of invasive basal diameters across the treatments. It also confirms the strong presence of the invasives on the restoration site.

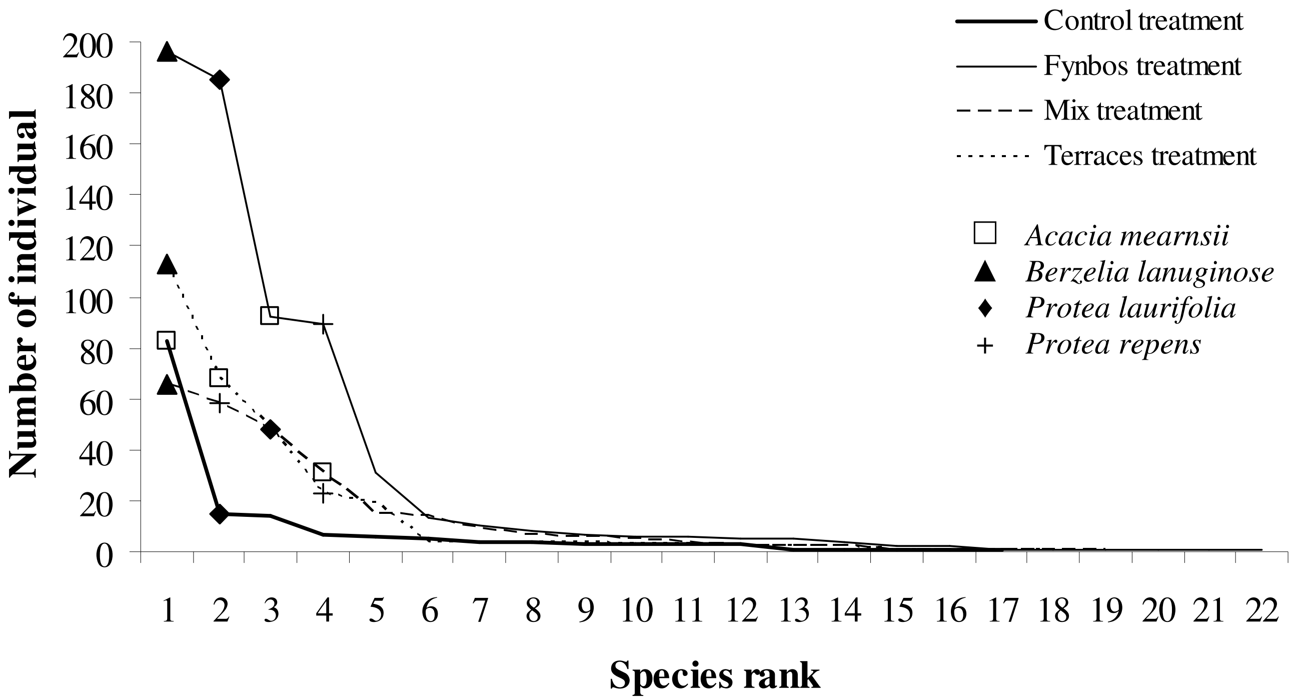


Fig. 2. A rank-abundance curve of the woody species within the 50 m^2 plots.

A clear dominance of a few woody species occurs on the Oaklands site as shown by Fig 2. The dominant species are *Berzelia lanuginosa* (L.) Brongn., *Protea laurifolia* Thunb., *Protea repens* (L.) L. and *Acacia mearnsii* De Wild. Fewer individuals of the woody indigenous fynbos species occur within the control treatment plots where no seeds were sown. The number of *Acacia mearnsii* De Wild within control treatment plots are almost the same as in the fynbos treatment plots. The fynbos treatment plots hosted a substantially larger number of dominant woody fynbos individuals compared to the other two sowing treatments. The fynbos treatment plots had the highest number of species present.

The size-class distribution of *Acacia mearnsii* De Wild. shows the typical pattern created by continuous recruitment (Fig. 3a) while a large number of small trees are emerging within the community. *Protea laurifolia* Thunb. (Fig 3b), *Protea repens* (L.) L. (Fig 3c) and *Berzelia lanuginosa* (L.) (Fig 3d) show event-driven recruitment which means that their populations recruit after a fire disturbance and are thus resilient to fire events. *Brabejum stellatifolium* L. also shows a faint pattern of event driven recruitment. These diagrams confirm the great absence of indigenous woody individuals within the control plots while the fynbos treatment plots have the largest number of indigenous individuals. It is also clear in Fig. 3 that the indigenous species do not occur abundantly (if at all) within the control plots.

Table 1 shows the means and standard deviations of all the woody species recorded and identified within the different treatment plots. It is useful to compare this analysis with the plots of Fig 3. None or only one specimen was recorded for some indigenous species listed and therefore there is no indication of a mean and standard deviation for those species. All the invasive tree species present on the restoration site are completely represented in the control plots. Only *Acacia mearnsii* De Wild. occurs within all treatments while *Eucalyptus grandis* W.Jill ex Maiden and *Hakea sericea* Schrad. & J.C. Wendl. occur in two other treatments respectively. *Berzelia lanuginosa* (L.) has a much larger mean basal diameter within the control treatment plots than within other treatment plots. However, it must be noted that there were only 4 individuals in the control plots while the other treatment plots had over 50 individuals (see Fig 3d). The fynbos treatment plots have the smallest basal diameter mean for *Berzelia lanuginosa* (L.) but also have the largest number of individuals (196) while

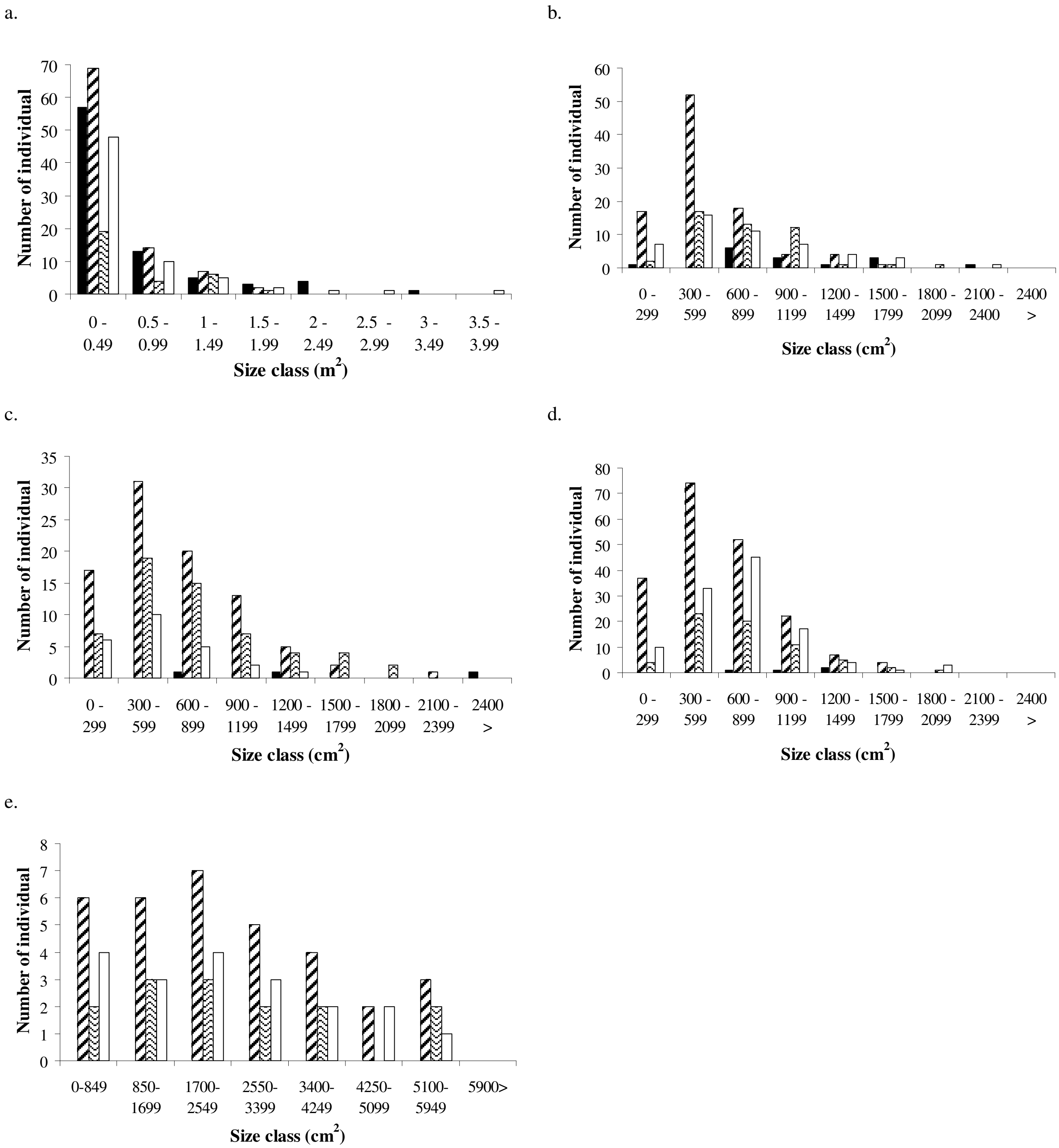


Fig. 3. The size-class distributions of *Acacia mearnsii* (a), *Protea laurifolia* (b), *Protea repens* (c), *Berzelia lanuginosa* (d) and *Brabejum stellatifolium* (e). Solid bars = control treatment, thick cross hatched bars = fynbos treatment, thin cross hatched bars = mix treatment, clear bars = terraces treatment.

the means within the mix and terraces plots are almost the same with no significant difference between them. There are significant differences in mean basal diameter for *Leucadendron salignum* P.J. Bergius. between the fynbos, mix and terraces treatments. The terraces treatment has the largest mean while the mix treatment has the smallest mean. *Protea laurifolia* Thunb. has the largest basal diameter mean within the control treatment plots which differ significantly from the means of the fynbos and terraces treatments. Again the lowest number of plants occurs within the control treatment plots (15). A significant difference was identified between the fynbos and mix treatments while the fynbos treatment also had a significantly larger number of individuals.

Species Name	Control treatment	Fynbos treatment	Mix treatment	Terraces treatment
Woody Invasive Species				
Acacia mearnsii De Wild.	5.83±4.30	5.10±4.25	7.04±4.43	5.75±4.36
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	2.04±0.82^{m,t}	2.48±0.81^{f}	2.39±0.93^{f}
Brabejum stellatifolium L.	-	5.54±3.13	6.62±3.83	6.47±3.13
Diospiros glabra (L.) De Winter	1.62±1.77	1.35±0.74	-	3.50±0.00
Leucadendron salignum P.J.Bergius	-	1.62±0.35^{m,t}	0.72±0.11^{f,t}	2.31±0.34^{f,m}
Leucadendron salisifolium (Salisb.) I.Williams	-	-	5.53±2.02	7.20±2.50
Maytenus oleoides (Lam.) Loes.	-	-	-	-
Metrosideros angustifolia (L.) J.E. Sm.	5.73±1.72	-	4.27±1.14	-
Myrsine africana L.	-	0.60±0.19	-	-
Olea europaea subsp. *africana*	-	-	-	-
Olea capensis L. subsp. *capensis*	-	-	-	-
Podalyria calyptrate (Retz.)Willd.	-	-	-	-
Protea laurifolia Thunb.	4.44±1.87^{f,t}	2.82±1.22^{c,m}	3.41±1.33^{f}	3.09±1.39^{c}
Protea repens (L.)L.	4.58±1.09	2.86±1.18	3.14±1.26	2.69±1.19
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

Superscripts indicate where significant differences exist between the treatment and the letter: c = control treatment, f = fynbos treatment, m = mix treatment and t = terraces treatment.

Table 1. The basal diameter means and standard deviations of the woody species within the different treatment plots.

Seedling survey

The rank-abundance curve of the seedling survey data (Fig. 4) shows that there is again a strong dominance of a few species with large numbers of individuals within the 1m^2 survey plots. A total number of 47 species were recorded and identified. *Anthospermum aethiopicum* L. that was used as mulch and as a seed source in the 1998 experiment is currently the most abundant species within the plots of three sowing treatments (control, mix and terraces). *Gnidia tomentosa* L., an indigenous perennial shrub that originated from the seed bank, is the second most abundant species within all sowing treatments. *Pteridium aquilinum* (L.) Kuhn subsp. *aquilinum* (Bracken) was the dominant ground cover under the acacia canopy prior to the clearing treatment

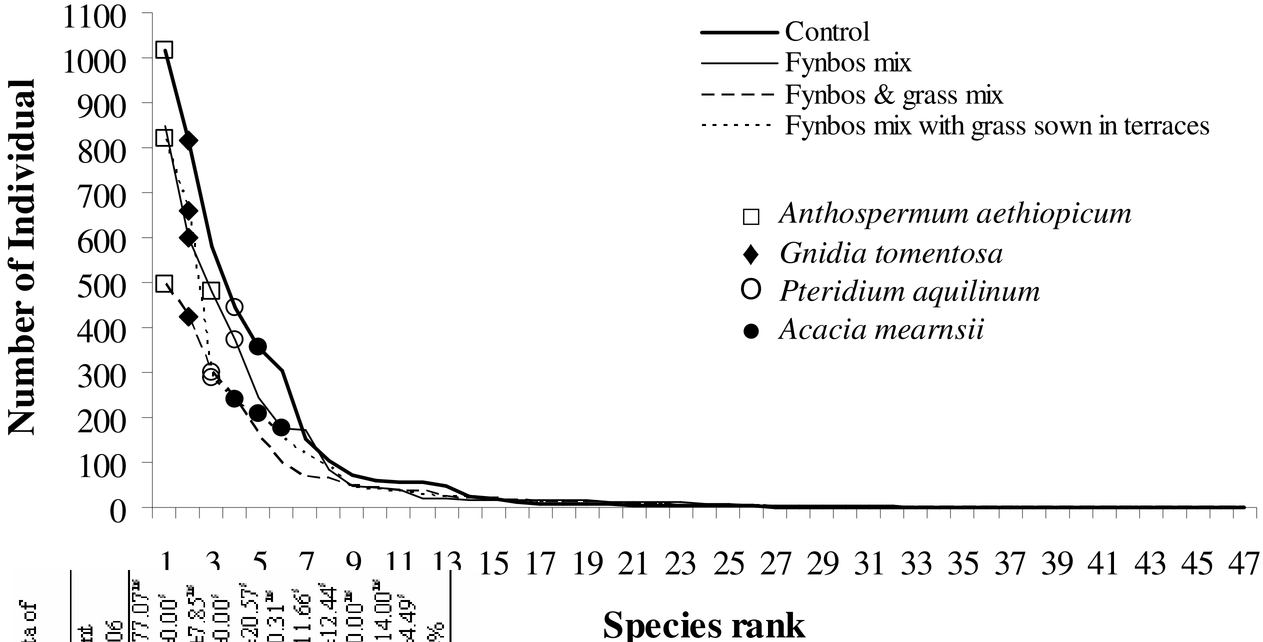


Fig. 4. A rank-abundance curve of the seedling survey.

Table 2. Mean density counts with standard deviations of each growth form guild within the four different sowing treatments. The 1998 data of the first vegetation census is compared with the 2006 re-assessment survey data.

Growth form guild	Control		Fynbos treatment		Mix treatment		Terraces treatment	
	1998	2006	1998	2006	1998	2006	1998	2006
Ericoids	42.6±52.47	50.95±65.45^{ns}	75.5±77.48	25.05±43.46^{s}	54.30±45.45	27.05±33.88^{s}	46.10±35.96	41.60±77.07^{ns}
Proteoids	0.05±0.23	0.00±0.00^{ns}	7.40±5.84	0.00±0.00^{s}	4.90±4.00	0.00±0.00^{s}	6.46±6.06	0.00±0.00^{s}
Acacia mearnsii	7.65±7.24	17.90±25.81^{s}	9.11±10.44	9.03±9.21^{ns}	9.86±8.00	11.95±12.85^{ns}	10.50±13.32	10.50±7.85^{ns}
Eucalyptus spp.	4.43±2.23	0.10±0.31^{s}	1.35±1.63	0.70±2.47^{ns}	0.95±1.32	0.00±0.00^{s}	1.20±1.24	0.00±0.00^{s}
Indigenous grasses	2.35±3.20	41.95±53.08^{s}	5.55±10.66	46.60±90.16^{s}	40.50±33.88	9.60±11.15^{s}	56.00±36.51	13.05±20.57^{s}
Alien grasses	0.15±0.67	0.40±0.68^{ns}	0.00±0.00	0.25±0.55^{ns}	57.50±61.63	0.05±0.22^{s}	17.00±45.20	0.10±0.31^{ns}
Indigenous forbs	2.35±3.13	3.55±5.65^{ns}	2.35±3.34	3.05±3.85^{ns}	1.80±2.21	4.60±8.39^{ns}	2.00±3.21	9.45±11.66^{s}
Alien forbs	1.00±1.45	21.80±54.47^{ns}	0.80±1.28	7.70±10.63^{s}	3.25±10.69	6.05±5.98^{ns}	0.65±1.42	10.40±12.44^{s}
Broadleaved shrubs	0.00±0.00	1.25±4.90^{ns}	0.05±0.22	19.35±17.24^{s}	0.00±0.00	0.15±0.37^{ns}	0.45±1.61	0.00±0.00^{ns}
Bracken	29.1±18.63	22.3±19.18^{ns}	37.0±25.17	18.70±17.20^{s}	20.20±20.25	15.10±15.60^{ns}	22.10±23.7	14.35±14.00^{ns}
Geophytes	0.40±0.75	2.3±3.15^{s}	0.70±1.42	15.45±31.41^{s}	0.20±0.52	3.90±10.11^{ns}	0.30±0.92	2.40±4.49^{s}
Total cover	35%	30%	42%	30%	37%	17%	31%	24%

ns = no significant difference between 1998 and 2006 data, s = significant difference between 1998 and 2006 data

and also was the first to emerge after the clearing treatment fire in 1998 (Prins, 2003). It is still one of the most abundant species on the site but by comparing the 1998 and 2006 seedling density data, it is clear that Bracken numbers reduced within all treatment plots while reducing significantly within the fynbos treatment plots. An alarming number of *Acacia mearnsii* De Wild. seedlings have emerged among the other seedlings with substantially more seedlings emerging in the control treatment plots. The lowest number of acacia seedlings is present within the fynbos treatment plots. By comparing the 1998 and 2006 seedling survey data (Table 2) it is clear that the number of acacia seedling recruits stayed approximately the same within the three sowing treatments but increased significantly in the control treatment.

The means and standard deviations of the density counts of seedlings within each sowing treatment are given in Table 2. The table compares data from the 1998 study with the re-assessment survey data of 2006. Ericoid density reduced significantly in the fynbos and mix treatment plots since 1998. Unfortunately, no protea seedlings were recorded in the re-assessment survey. The density of *Eucalyptus grandis* W.Jill ex Maiden decreased in every sowing treatment with their numbers decreasing significantly in the control, mix and terraces plots. Indigenous grass density increased significantly within the control and fynbos treatment plots while it decreased significantly within the mix and terraces treatment plots. The density of the alien grasses that were sown was low in both the 1998 and 2006 surveys except for being abundant within the mix treatment plots in 1998 which have reduced significantly in 2006. Forb density remained stable except for alien forb increases within fynbos and terraces treatment plots and indigenous forb increases within terraces treatment plots.

Broadleaved shrubs increased significantly in fynbos treatment plots while decreasing to zero in terraces treatment plots. The broadleaved shrubs surveyed in the 2006 recruitment survey were actually not seedlings surveyed but resprouting adult plants or sprouting shoots from adult plants. Geophyte density increased significantly in the control, fynbos and terraces plots. The total cover percentages shown at the bottom of Table 2 indicate that the total cover estimated in 1998 was generally higher than what was estimated in 2006.

Discussion

Eight years after initial clearing *Acacia mearnsii* De Wild. still has a strong presence and remains one of the dominant species on the restoration site. This is because the site did not receive any follow-up clearing after it has been initially cleared in 1998 by the Working for Water teams. However, by comparing the control plots with the plots sown with indigenous seed and grass seed mixes, there is a substantial difference in the presence and abundance of indigenous fynbos species. The plots that received the sowing treatments had substantial populations of indigenous species. The presence of a small number of indigenous woody plants (an average of 11.8 plants per plot) within the control treatment plots could be the result of seed dispersal from adult plants in adjacent plots or they may have recruited from a seed bank.

The fact that there is a smaller number of woody plants present within the control plots may be a reason for the larger basal diameter mean of woody indigenous plants compared to the basal diameter means within the other sowing treatments. Nutrients may be more readily available because of less competition. The equally sized basal diameter means of the woody invasive trees within all sowing treatment plots indicates that the acacias are able to obtain nutrients sufficiently with or without competition from other plants. *Acacia mearnsii* De Wild., amongst other invasive trees, possess competitive abilities to enhance nutrient acquisition and obtaining scarce resources through mechanisms of nitrogen fixing symbionts, sheathing mycorrhizas and extensive root production (Stock and Allsopp, 1992)

Results of the re-assessment survey show that restoring the site after alien clearing by sowing indigenous seeds did increase the diversity on the site by improving species presence and abundance. This was also found in the 1998 seedling survey (Prins, 2003). However, the 2006 results also show that a few species that were sown currently dominate the restoration site which indicates that the indigenous diversity could not be fully restored and that species abundance is unevenly distributed among species. A seed mix including only fynbos and riparian species appear to improve diversity more efficiently. However, the inclusion of grasses seems to have suppressed the presence of alien invasive acacias to some degree. By revegetating a site after invasive alien clearing the reinfestation of a site can be avoided (Holmes et al., 2005).

The re-established community seems to be resilient to fire disturbances. Three fynbos species show event driven recruitment while most resprouting species within the plots resprouted after the 2005 summer fire. However, *Acacia mearnsii* still co-dominates the site as it is also resilient to fire disturbance and shows continuous recruitment. Also, by comparing the seedling density data of 1998 and 2006 it appears that the total seedling density of the community increased which is also an indication of the community's growth progress and resilience to fire.

Some interesting findings were made by comparing the 1998 and 2006 density counts of seedlings on the restoration site. Ericoid seedlings reduced significantly in numbers. The reason for this result may be that a number of plants could only be identified as far as to say that they are indigenous shrubs. They could have been listed under Ericoid shrubs during the 1998 study.

The absence of protea seedlings on the site remains largely unexplainable. The two protea species present on the restoration site are both serotinous species which means that they store their seeds above the ground in cones. Such species are killed by fire disturbances and regenerate only from the seeds stored in the canopy (Bond, Vlok and Viviers, 1984; Bond, Maze and Desmet, 1995). Cold, wet weather with fluctuating temperatures is cues for most protea seeds to germinate (le Maitre and Midgley, 1992). However, if drought follows good rains, the seeds of serotonous species may fail to recruit because they germinate readily once they are released (Zammit and Westoby, 1987). Bond et al. 1984 state that the recruitment of serotinous proteas is confined to the period immediately after disturbance and is negligible at other times. *Protea repens* has been found to have a moderate regeneration after summer fires (Bond, Vlok and Viviers, 1984). Proteaceae seeds are short-lived after the release from the parents and do not persist to form reserves after the death of the parent (Bond, Vlok and Viviers, 1984). Local serotonous populations can become extinct because of a fire disturbance before they are reproductively mature or as a result of no fire disturbances being experienced during the lifespan of the species (Bond, Vlok and Viviers, 1984). It is not known whether fire disturbances occurred between the 1998 and 2005 fire events. However, many pre-mature protea plants were present on the site that was killed by the 2005 summer fire. The presence of adult and young protea plants on the site may be proof of at least one fire disturbance between the 1998 and 2005 period that led to the regeneration of these younger plants. Proteas mature at about four years of age (le Maitre and Midgley, 1992) so if the fire occurred round about 2003-04, the sowed protea recruits would have been old enough to regenerate after the fire. However, the occurrence of another fire in 2005 would have killed all the new recruits.

When comparing the 2006 and 1998 seedling density data, it appears that sowing a mixture of indigenous seed as well as including alien grass seed appears to prevent the increase in number of *Acacia mearnsii* De Wild. seedling recruitment. The fact that the whole tributary is still invaded by acacias means that acacia seeds may have dispersed onto the plots from the surrounding area which makes the findings of the survey even more interesting. While *Acacia mearnsii* De Wild. is constantly releasing seeds, the indigenous vegetation in the seeded plots still prevent them from recruiting. Introducing seed mixes also seems to significantly reduce the number of *Eucalyptus grandis* W.Jill ex Maiden and *Pteridium aquilinum* (L.) Kuhn subsp. *aquilinum* seedling recruits. The decrease in density of *Pteridium aquilinum* (L.) Kuhn subsp. *aquilinum* means that other seedlings have a chance to recruit because the dominance of *Pteridium aquilinum* (L.) Kuhn subsp. *aquilinum* could suppress the regeneration of other fynbos plants (Prins, 2003).

It is interesting that the indigenous grass density significantly reduced in the plots where the alien grasses were sown while increasing where only fynbos seeds were sown. However, the alien grass density also decreased in the plots where they were sown and increased only slightly in the control and fynbos treatment plots. 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 was predicted by the 1998 consecutive survey data.

An explanation for the total cover percentage differences between 1998 and 2006 may be that of differences in judgement between the two different recorders. Or it may be the fact that the survey in 1998 took place two and a half weeks later in the season after the winter rains than the survey done in 2006 which resulted in the seedlings having more growth time.

Conclusion

The Oaklands re-assessment survey showed that using indigenous seed mixes, a fire resilient indigenous community can be re-established which increases the area's species richness and also increase the abundance of some species. It also showed that the re-established indigenous vegetation cover reduced the recruitment of *Acacia mearnsii* De Wild. seedlings. However, more research is needed about ecosystem repair after alien clearing (Richardson and van Wilgen, 2004). Sparsely invaded or short term densely invaded ecosystems do not need further management interventions and can recover after alien clearing (Richardson and van Wilgen, 2004). However, some ecosystems can not recover after alien clearing and need to be restored to some desirable condition, may it be a certain indigenous species composition or ecological function. The causes of degradation must be dealt with first to ensure realistic restoration goals and success with the restoration project (Stewardson et al., 2004). The cause of degradation was not completely dealt with in the tributary on Oaklands Farm therefore the site could not be fully restored. It is very important that more knowledge is gained about optimal methods of recovering an area's native vegetation cover after alien plant clearing (Macdonald, 2004). To actively rehabilitate an area after clearing is an expensive operation while the results remain uncertain and local area biodiversity loss is likely to be high (Macdonald, 2004). The best option is to prevent invasion in the first place (Macdonald, 2004).

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