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Assessment of post-burn removal methods for *Acacia saligna* in Cape Flats Sand Fynbos, with consideration of indigenous plant recovery

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ABSTRACT

The Greater Cape Floristic Region (GCFR) of South Africa is a biodiversity hotspot threatened by the impacts of habitat transformation and invasive alien species. Cape Flats Sand Fynbos (CFSF) is a critically endangered vegetation type occurring within the GCFR, and its largest remaining fragment is the focus of a large-scale invasive plant control and biodiversity restoration project. *Acacia saligna* is a highly problematic invasive in CFSF and the main target of the control. To mitigate damage caused by this species, stands are removed and burned, which stimulates both the large invasive seed-bank and the indigenous seed-bank in the soil. Although there are no clear methods on how to manage the re-invasion at this stage without damaging indigenous plant recovery, three post-burn removal methods have the potential to be effective: (1) cutting the *Acacia* saplings below the coppicing point, (2) cutting the saplings and applying herbicide to the stumps, and (3) foliar herbicide spray. The aims of this study were to (i) find the most effective post-burn *A. saligna* control treatment, (ii) find the treatment that causes the least harm to indigenous plant recovery, (iii) determine the most cost-effective treatment, and (iv) establish which treatment is the most suitable for large-scale use. Cutting below the coppicing point of the *A. saligna* provided the most effective removal and was also the least damaging to indigenous vegetation recovery. The foliar spray treatment, however, saved the most time and costs. The best method is therefore dependent on the project goals, scale, and density of the *A. saligna* invasion. These results may be applicable to other types of fynbos and to other fire-stimulated invasive *Acacia* species.

1. Introduction

Invasions by non-indigenous species have become a global issue; they are a leading cause of biodiversity loss, drive habitat degradation, and affect functional processes that provide ecosystem services (D'Antonio et al., 2001). Warm-temperate and sub-tropical areas account for much of the world's biodiversity but are prone to invasions (Holmes and Cowling, 1997). One such area is the Cape Floristic Region (GGCFR) in South Africa which is a botanical biodiversity hotspot severely threatened by invasive species (MacDonald and Richardson, 1986; Van Wilgen et al., 2001). Fynbos, a fire-driven, low-shrub vegetation biome prevalent in the GCFR (Cowling et al., 1992), is also the most impacted and sensitive biome of the region (Richardson et al., 1992). Unfortunately, much of the biodiversity in the GCFR is not adequately protected or managed, particularly those vegetation types adjoining areas of development and urbanization, such as the Cape Flats Sand Fynbos (CFSF) (Rebelo et al., 2006, 2011). The CFSF has high species richness typical of fynbos (Cowling et al., 1992; Rebelo et al., 2006) with several endemics (Holmes et al., 2008) and occurs within the limits of the city of Cape Town. Urban development has transformed 85% of the CFSF and only 1.5% is protected (Rebelo et al., 2011). The goal set to protect 30% for the CFSF is now unattainable (Rebelo et al., 2011), and the few remaining areas are mostly degraded and invaded by alien species. Thus, CFSF is categorized as a critically endangered vegetation type, and restoration is required to prevent its extinction (Rebelo et al., 2011).

Acacia saligna (Labill.) H.L.Wendl was introduced to South Africa to anchor unstable sand dunes (Shaughnessy, 1980) and is now one of the most problematic invaders of the GCFR. *A. saligna* not only outcompetes indigenous plant species by growing faster and taller, but it also transforms the environment by creating shady canopy cover and by altering soil properties through a combination of fixing nitrogen and its high input of leaf litter (Witkowski, 1991; Holmes and Cowling, 1997). These changes inhibit growth of indigenous species and alter the structure of the original vegetation, facilitating the spread of weedy species and grasses (Holmes and Cowling, 1997).

Altering of soil properties is a particular issue in the GCFR where much of the vegetation has adapted to grow in relatively nutrient

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poor soils (Moll and Jarman, 1984; Lamb and Klaussner, 1988). Natural fire regimes in the GCFR favor *A. saligna* recruitment. *A. saligna* seeds are produced in abundance and can remain dormant in the soil for over 50 years (Holmes et al., 1987; Richardson and Kluge, 2008). Germination occurs after the hard seed coat is damaged enough, for example by heat, to allow for water penetration (Milton and Hall, 1981; Jeffery et al., 1988). After fire, seeds can successfully germinate from soil depths up to 10 cm, quickly outcompeting indigenous re-growth, and increasing the density of the acacia stand (Holmes and Cowling, 1997; Strydom et al., 2012). *A. saligna* also has the ability to re-sprout if cut, from auxiliary buds left on the stem or from a meristematic zone where the stem transitions to the tap root, the 'coppicing zone' (MacDonald and Wissel, 1992; Love et al., 2009). The combination of impacts from a dense *A. saligna* invasion can cause major shifts in both ecosystem structure and function, which can be challenging to reverse (Yelenik et al., 2004; Hobbs et al., 2009). Efforts have been made to improve our understanding of the invasion process of *A. saligna* in fynbos and to determine effective removal methods while considering natural vegetation recovery in relation to diversity and structure (Arim et al., 2006; Le Maître et al., 2011). Removal of mature stands is usually accomplished by cutting and applying herbicide to cut wounds, or by foliar application of herbicide (MacDonald and Wissel, 1992). Mere removal of the invasive trees, however, does not ensure that indigenous vegetation will recover and does not address the long-lived alien seed banks remaining in the soil (Holmes et al., 1987; Zavaleta et al., 2001). In response to this issue, it has been recommended to burn invasive biomass following cutting (Holmes et al., 1987), while maintaining a close to natural fire intensity to avoid damaging indigenous seeds in the soil (Richardson and Kluge, 2008). This method not only reduces the nitrogen levels of the soil and stimulates the indigenous seed bank (Le Maître et al., 2011) but also causes a mass germination of *Acacia* propagules, which can reduce the invasive seed bank by 90% in the first year (Holmes et al., 1987).

It is generally acknowledged that the invasive re-growth post fire must receive continued treatment to avoid maturation and regeneration of the *A. saligna* stand, but there is little information regarding the best method of control at this point (Le Maître et al., 2011). Non-specific area-wide control measures, such as bulldozing or large-scale foliar spray, are often implemented due to the extreme density of the germinating *Acacia* (MacDonald et al., 1985). However, as indigenous vegetation recovery also begins after burning, it is important to define methods that are the least damaging to natural regrowth (Le Maître et al., 2011). Numerous methods of *Acacia* removal have been tested. Of these, cutting combined with herbicide application or foliar spray have the potential to be effective control options after burning. However, high densities of re-growing *Acacia* make applying herbicide to stumps or spraying leaves potentially time consuming and costly (Van Wilgen et al., 2001), while herbicide contamination of soil and non-target species can negatively impact indigenous vegetation recovery (Souza-Alonso et al., 2013). Another possible control treatment is one utilized for removal of another highly invasive species *Lantana camara*. Like *A. saligna*, *L. camara* has strong re-sprouting and coppicing capabilities, yet eradication has been successful without chemicals by cutting the plants under the soil surface below the meristematic coppicing zone, paired with seed bank control (Love et al., 2009). This method is known as the 'cut rootstock method' and could potentially be utilized for the control of *A. saligna* re-growth. The most effective method for post-burn removal of *A. saligna* ultimately depends on a variety of factors including its effectiveness, potential impact on indigenous biodiversity, and cost. We compared three methods of *A. saligna* re-growth removal in the Cape Flats Sand Fynbos within 18 months after burning the felled *A. saligna* stand in a control operation: (i) cutting with herbicide application to stumps and hand-pulling of seedlings ('cut and poison'), (ii) cutting the rootstock below the surface and hand-pulling seedlings ('cut rootstock'), and (iii) foliar spray herbicide application ('foliar spray').

We evaluated each method in terms of their (i) effectiveness in eradicating *A. saligna* regrowth, (ii) destructiveness to indigenous plants, (iii) cost-effectiveness, and (iv) suitability for large scale restoration after *A. saligna* removal.

2. Methods

2.1. Study site

This study was conducted at Blaauwberg Nature Reserve, Cape Town, in densely alien-invaded Cape Flats Sand Fynbos (CFSF), a vegetation type located on the acidic sands in the lowlands of Cape Town. In its pristine condition, this medium height shrubland is mainly composed of species of Proteaceae, Restionaceae, Ericaceae, and Asteraceae (Rebelo et al., 2006). The prevailing climate is Mediterranean in character with cool winter rainfall (575 mm annual average) falling predominantly between May and August, contrasted by hot, dry summers.

As restoration resources are limited and often focused on less severely degraded areas, this dense stand of *A. saligna* would not be a priority for clearance were it outside a protected area or in non-endangered vegetation (Hobbs, 2007; Reid et al., 2009). Invasion severity and recovery potential were mapped throughout the area via pre-removal surveys which included *A. saligna* density and density of indigenous species as variables. Removal of the *A. saligna* stand was conducted in a 32 ha portion of the reserve by cutting and applying herbicide to the stumps, followed by a dry season prescribed burn. A biological control agent is also present in the area, the gall-forming fungus *Uromycladium tepperianum*, which slows the growth of the acacia and reduces its lifespan (Morris, 1997).

2.2. Experimental design

This study assessed an area where the felled acacia slash was spread and 'block burned' in early April 2013. Ten, separate treatment blocks were set up across this 32 ha area in August 2013. Pre-clearance treatment block restoration potentials ranged from low (1–10% indigenous cover) to very low (<1% indigenous cover) (Appendix A). Each treatment block consisted of four 5×5 m plots, representing three treatments and a control (Fig. 1), resulting in a total of 40 plots.

Post-fire surveys were divided between pre-treatment (last week of August to September 2013) and post-treatment (last week of August 2014). Survey dates were chosen to coincide with early spring, when seedlings emerge in response to winter rainfall. We measured the following variables at each plot during surveys: (1) percentage cover of *A. saligna* (2) percentage cover of each plant species, indigenous and alien (3) density of *A. saligna* and (4) density of each plant species other than *A. saligna*. During the first week of January 2014, a second pre-treatment survey was done to assess density-dependent mortality and growth in *A. saligna*. Two variables were sampled: (1) percentage cover of *A. saligna* and (2) density of *A. saligna*. The percentage cover and density of *A. saligna* were recorded to determine the effectiveness of each treatment in the removal of the post-fire re-growth. The percentage cover and density data for all other plant species were recorded to determine the impact of each treatment on indigenous vegetation recovery.

The estimated percentage cover of each plant species including *A. saligna* was determined visually. The density of the *A. saligna* re-growth was recorded by using a 1×1 m quadrat, divided into twenty-five 20×20 cm subsections. The quadrat was placed towards the center of each plot and the total number of *A. saligna* plants found in 10 of the subsections counted. Subsections were chosen at random. This count was used to determine the average density of *A. saligna* per square meter. The density of each plant species excluding *A. saligna* was determined for all individually growing plants by counting each individual.

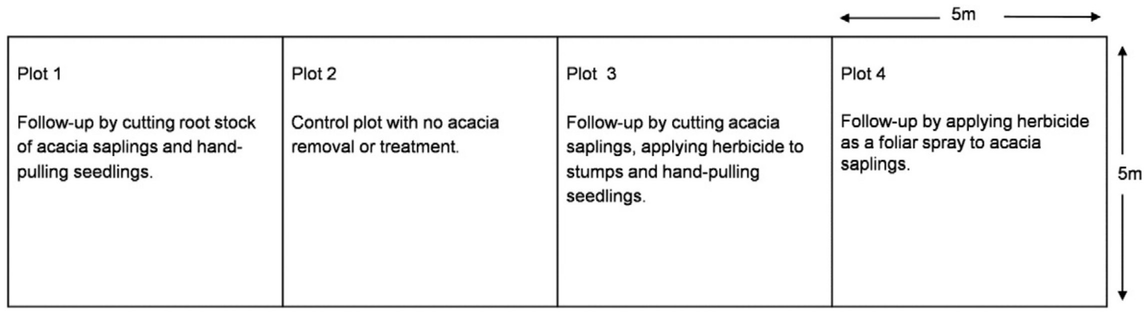

Fig. 1. An example of a treatment block layout at the BNR study site for the different post-burn *A. saligna* removal methods. The block is sub-divided into four 5 × 5 m plots, each with their corresponding treatment or control.

Colony-forming species, like some grasses, could only be recorded utilizing percentage cover. Plants were identified to species level where possible and to distinguish between indigenous and alien species. An alien species was regarded as any plant species that did not naturally occur in the CFSF (Manning, 2007; Bromilow, 2010).

Treatments were applied approximately 1 year after burning, from mid-March to mid-April of 2014. For the 'cut rootstock' and 'cut and poison' treatments, all cutting of the *A. saligna* regrowth was done manually using large loppers. To cut off the rootstock, the loppers were pushed into the sand aiming toward the plant stalk at roughly 3–5 cm in depth. For stump poisoning, a cut stem was left above ground to which herbicide was then immediately applied using a small handheld sprayer. These first two treatments were then followed with hand-pulling of any remaining *A. saligna* seedlings. For the 'foliar spray' treatment, buckets were placed over any indigenous shrubs and shrublets before the *A. saligna* foliage was lightly coated with herbicide using a back-pack sprayer. The herbicide utilized for both the stump treatment and foliar spray was labeled as *Confront Super* (active ingredients Triclopyr 120 g/l, Aminopyralid 12 g/l); diluted at a ratio of 70 mL herbicide in 10 L water with 1 tablespoon of blue dye added to indicate treated plants. The herbicide costs R128.80 per liter (in 2014). During all surveying and treatments, we attempted not to damage indigenous plants or control plots. The time taken to conduct each of the treatments was also recorded for half of the sampling blocks.

2.3. Data analysis

For all statistical tests, mixed-model repeated-measures ANOVA was used with plot number as random effect and treatment and time as fixed effects. Normality assumption was checked by inspecting the normal probability plot and was found to be acceptable. Fisher's LSD post hoc test was used to assess significant differences within treatments and before and after treatments. All statistical analysis were conducted in STATISTICA 11 (Statsoft Inc, 2013).

For analysis of *A. saligna* density and percentage cover each plot was regarded as an individual replicate, giving 10 replicates per treatment. The dependent variables were analyzed separately to compare the effects of the three treatments and the control. This was done both before and after the treatments were applied to ensure that any significant results found among treatments were not because of initial natural variation. Each of the dependent variables were also analyzed separately for each individual treatment to compare the plots before and after the treatment was applied. For *A. saligna* cover and density, this analysis included the two pre-treatment and one post-treatment survey to ensure that any significant results from the treatments were not due to intra-specific competition or impacts of the biocontrol fungus.

Density of each plant species recorded excluding *A. saligna* was used to calculate diversity indices for each plot. The Simpson's Index of Diversity (1-D) (Peet, 1974) and the Shannon Diversity Index (H) (Zar, 2010) were calculated, and as described above, the diversity indices were first compared between the treatments and control, and then before and after for each individual treatment. Diversity comparisons were limited to indigenous species only.

All recorded plant species except *A. saligna* were allocated to one of the following growth forms; shrub, shrublet, forb, geophyte, graminoid, alien graminoid, or alien (non-graminoid). The percentage cover of each growth form in each plot was then determined by adding the percentage cover of each constituent plant species.

Species composition was firstly assessed by plotting a rank-abundance curve for each of the treatments and the control using the list of species recorded during the post-treatment survey. Secondly, the species density data for each plot were square root transformed to comply with the assumption of normality. A Bray Curtis Similarity analysis was then used to plot an MDS of the similarity in species composition between the treatments and the control. Primer 6 was used to transform the data and plot the MDS.

3. Results

We identified 66 indigenous plant species from 25 families; 36 species were recorded before treatments, and 55 post-treatment (Appendix B). No species endemic to CSFS were found (Rebelo et al., 2006). In addition, we found 18 alien species (*A. saligna* excepted) in the treatment plots (Appendix C). Plots treated using 'cut rootstock' and 'cut and poison' methods were almost devoid of living *A. saligna* cover, while plots using 'foliar spray' treatment had low live cover but a high amount of standing dead and leaf litter. Implementation of the 'cut rootstock' and 'cut and poison' treatments required approximately 4 person hours per plot, 16 times longer than the 'foliar spray' treatment which required 0.25 person hours per plot. The quantity of herbicide used in the herbicide treatments 'cut and poison' and 'foliar spray' was not calibrated to a specific quantity per plot or per plant, thus the amount used per treatment could not be determined.

3.1. Effects of treatments on invasive Acacia

Acacia percentage cover and density did not differ significantly between treatment and control plots before treatments were applied. Four months after treatments were applied, acacia cover (and density) were significantly lower compared to the control (cut root stock $p < 0.00000$; cut poison $p < 0.00000$; foliar spray $p < 0.00000$).

A. saligna cover and density did not differ between the removal treatments. When *A. saligna* cover was compared between the initial and final surveys for each treatment, it was found that all three removal methods resulted in significantly lower *Acacia* cover ('cut root stock' $p < 0.000063$; 'cut and poison' $p < 0.000019$; 'foliar spray' $p < 0.000336$). The control plots showed that *A. saligna* cover increased significantly without treatments ($p < 0.00000$). The 'cut rootstock' treatment had the lowest average *A. saligna* cover, and the least variation. When comparing *A. saligna* density from the initial to final survey, densities were significantly lower for all removal treatments and the control post-clearing ('cut root stock', $p < 0.00000$; 'cut and poison', $p < 0.00000$; 'foliar spray', $p < 0.00000$; 'control', $p < 0.0009989$).

3.2. Effects of treatments on indigenous vegetation parameters

Before treatments were applied, the variation among indigenous species diversity was high. There was no significant difference in the Simpson's or Shannon diversity indices between treatment plots before *A. saligna* removal was implemented.

After removal treatments were implemented, only the 'cut rootstock' method resulted in significantly higher indigenous species diversity for Simpson's index than the control ($p < 0.011721$). For the Shannon index, all three treatments resulted in significantly higher diversity compared to the control (cut root stock $p < 0.000220$; cut and poison $p < 0.003918$; foliar spray $p < 0.003010$).

All *A. saligna* removal treatments resulted in significantly higher species diversity in the final survey, 4 months after treatment, compared to the initial pre-treatment survey (Simpson's: 'cut root stock' $p < 0.019031$; 'cut and poison' ns; 'foliar spray' $p < 0.002588$; Shannon: 'cut root stock' $p < 0.000099$; 'cut and poison' $p < 0.002351$; 'foliar spray' $p < 0.000247$).

The percentage cover of each growth form did not differ significantly between the removal treatment and control plots in the pre-treatment survey. However, 4 months after treatment, compared to control plots, overall percentage cover was greater where cut root stock treatment was implemented (Fig. 2). For indigenous cover, this pattern was driven by forbs ($p < 0.000010$), shrublets ($p < 0.000000$), and graminoids ($p < 0.018$). Comparison between control and 'foliar spray' plots and control and 'cut and poison' plots yielded significant increases in forb ($p < 0.0167$, $p < 0.002378$, respectively) and shrublet cover ($p < 0.0134$, $p < 0.00619$, respectively) (Fig. 2).

When comparing pre- and post-treatments, native graminoids were significantly higher in cover across all treatments including control ('cut root stock' $p < 0.000014$; 'control' $p < 0.0012$; 'cut and poison' $p < 0.000068$; 'foliar spray' $p < 0.007592$). Shrubs were significantly higher in 'control' ($p < 0.02068$), 'cut root stock' ($p < 0.001495$), and 'cut and poison treatments' ($p < 0.021212$), while shrublets were significantly higher in all treatments but not in the control ('cut root stock' $p < 0.0000$; 'cut and poison' $p < 0.000619$; 'foliar spray' $p < 0.018713$).

3.3. Effects of treatments on species composition

The rank abundance curves plotted for the indigenous species in all plots 4 months following implementation of the treatments are depicted in Fig. 3. The 'foliar spray' treatment resulted in a relatively high dominance of a few species, shown by the high peak at the beginning of the curve. The dominant species were *Oxalis luteola*, *Isolepis* sp., and *Ficinia* sp., respectively. The 'cut root stock' treatment had a high representation of rare species, evidenced by the long tail of the curve. Without *A. saligna* removal, there was a more even dominance by the same three species as in the 'foliar spray' treatment, indicated by the rounded peak at the beginning of the control curve followed by a sharp drop. The control plots also resulted in an absence of rare species, shown by the short tail.

The MDS plot for the indigenous species composition of all plots 4 months after implementation of the treatments is shown in Fig. 4. The clumped distribution of the 'cut rootstock' treatment plots shows that the species composition resulting from this removal method is highly similar for each of the plots. The separation of this clump from the other treatments and control also indicates that the resulting composition is unique in comparison. The plots from the two herbicide treatments 'cut and poison' and 'foliar spray' are relatively grouped together with some overlap, indicating a similar outcome in species composition. Without *A. saligna* removal, there is higher variation in species composition indicated by the scattered dispersal of the control plots.

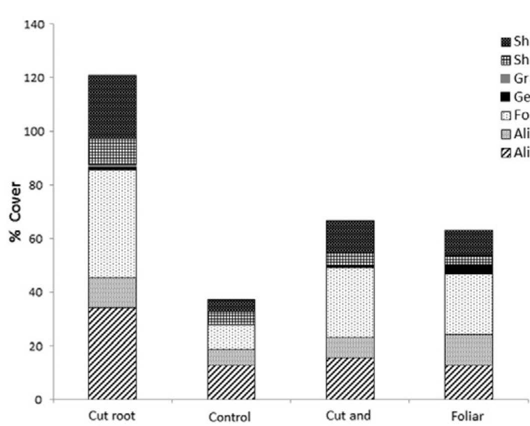

Fig. 2. The percentage cover of vegetative growth forms in the *A. saligna* removal plots at the BNR restoration project four months after treatment implementation.

4. Discussion

4.1. Effects of treatments on invasive Acacia

The commonly used removal treatments 'cut and poison' and 'foliar spray' were effective in reducing both the recruitment and re-growth of post-burn *A. saligna* (as measured by density and percentage cover). In addition, use of the 'cut root stock' removal method, relatively new for *A. saligna*, was successful. The best and most consistent removal resulted from this treatment, perhaps because of the simplicity of the method; the 'cut root stock' method effectively requires removal of whole plants below the meristematic coppicing zone, so none can be missed and no further herbicide application is required. Unlike 'cut root stock' removal, it was difficult to confirm that all plants had been properly treated using the two commonly used treatments, particularly when poisoning stumps which are dense and small and are thus hard to see. This may account for large variation in the results. This is not listed as an issue in a previous study, which was limited to the removal of mature, large stumped, *A. saligna* only at a much lower density (MacDonald and Wissel, 1992). The hand-held sprayers used to apply herbicide to the cut stumps did not indicate calibrated quantity; therefore, the amount applied to each stump was inconsistent. Similarly, variation in the 'foliar spray' treatment may also be due to a lack of dosage control. Absence of dosage control was likely exacerbated by the poor visibility of the green dye on the foliage that was designed for visibility on stumps (MacDonald and Wissel, 1992).

The post-burn reduction in *A. saligna* density over time, even without a removal treatment, may be due to intraspecific competition, effects of the biocontrol agent *U. tepperianum*, or a combination of the two. It can, however, be assumed that the *A. saligna* invasion without follow-up treatment is likely to be more severe in comparison to pre-burn (see Holmes and Cowling, 1997; Le Maître et al., 2011). A 90% mortality rate was recorded in the post-burn re-growth of a similar invasive, *Acacia mearnsii*, which reduced the density of seedlings but still resulted in a denser stand of *Acacia* than before the area was burned (Pieterse and Boucher, 1997). The results found by Pieterse and Boucher in 1997 are paralleled here in the control plots with reduced density yet significantly higher cover of *A. saligna*.

4.2. Effects of treatments on indigenous vegetation parameters

Without treatment, *A. saligna* seedlings that emerged after burning grew faster and taller than the indigenous plants, in keeping with the findings of Holmes and Cowling (1997). It is therefore crucial that re-growth is treated to continue *A. saligna* control and encourage fynbos recovery.

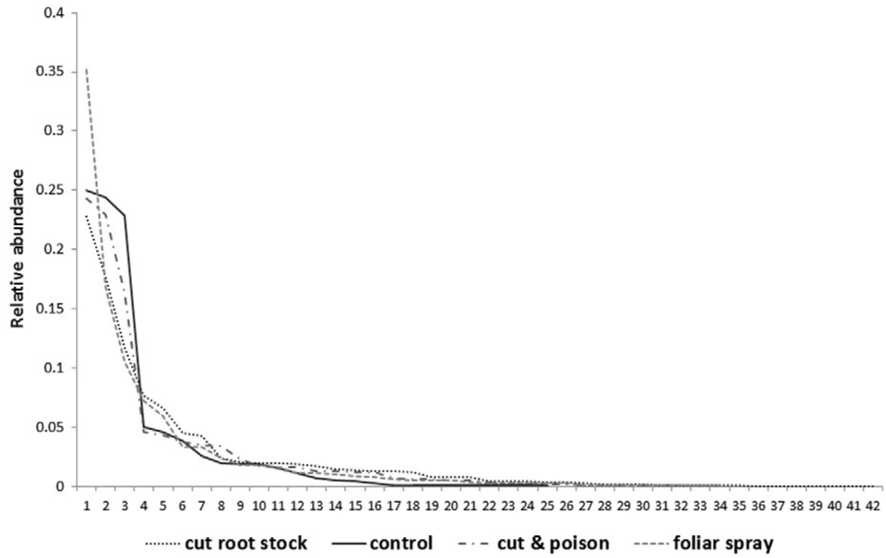

Fig. 3. The rank-abundance curve represents the post-treatment plant species composition in the experimental plots at the BNR restoration project. Curves are plotted for each removal treatment and the control. Only indigenous species were included in this analysis.

All three of the removal treatments tested led to an increased diversity of indigenous plants, probably due to the removal of interspecific competition for sunlight from *A. saligna* (see Cowling and Gxaba, 1990). The increase in indigenous species diversity supports related findings that fynbos species can regenerate from rootstock, geophytic structures and even long-lived soil-stored propagules (Holmes and Cowling, 1997).

When compared to the controls, only the 'cut rootstock' treatment resulted in significantly higher indigenous cover, which is important for conservation and has the additional benefit where increased cover of indigenous species aids in soil protection and can improve resilience to re-invasion (Holmes et al., 2000). Higher shrublet cover following any *A. saligna* removal, and significantly higher shrublet cover in the 'cut rootstock' plots compared to controls, suggests that species of this growth form are highly sensitive to the competition and shading from *Acacia*. Foliar spray herbicide application is highlighted as a potential negative issue in the structural recovery of the vegetation as only the 'cut rootstock' treatment, 'cut and poison treatment' and control plots showed an increase in shrub cover over the course of the study. It is clear that foliar spray treatments and the control somehow inhibit the recovery of one or more functional growth forms in comparison to cutting the rootstock of the *A. saligna*.

Follow-up treatments utilizing herbicide may encourage a dominance of grass species following *Acacia* removal as discussed by Le Maître et al. (2011). However, the results of this study show a significant increase in both indigenous and alien graminoid cover in all of the treatment and control plots, regardless of herbicide use. The success of fast-growing pioneer species following the removal of an invasive species is commonly reported and is related to the ecological transformations from the initial invasion benefitting these species, specifically the elevated nitrogen levels in the soil (Yelenik et al., 2004; Gaertner et al., 2012). The spread of grasses in the restoration site should be monitored as it has been found to suppress the recruitment and emergence of indigenous fynbos species (Holmes, 2008).

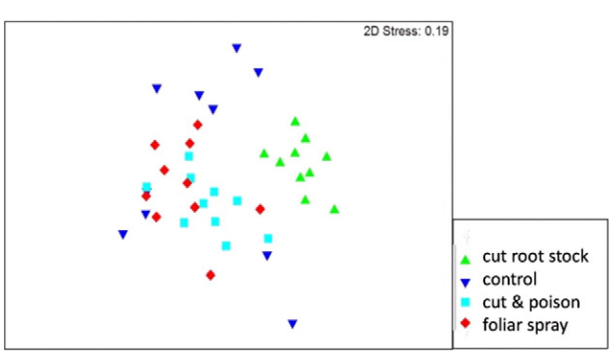

Fig. 4. The post-treatment indigenous species composition of the treatment and control plots from the BNR restoration project are represented in an MDS plot.

4.3. Effects of treatments on species composition

The dominance of few species (*Oxalis luteola*, *Isolepis sp.* and *Ficinia sp.*) in the plant assemblage was particularly strong following the 'foliar spray' removal treatment, possibly because wide coverage of herbicide during foliar spraying is potentially more damaging to non-target species. Because the herbicide treated foliage was shed onto the soil surface, it is also possible that more of the herbicide contaminated the soil and germinating plants. In the plots following the 'cut root stock' removal treatment there was a much higher representation of fynbos species represented by a low number of individuals. Impacts of herbicide treatment on certain fynbos species is again highlighted as an issue (see Souza-Alonso et al., 2013). These impacts may be less than those of regenerating *A. saligna*, however, as control plots had the poorest representation of represented by a low number of individuals species.

The combination of dominant herbaceous species such as *Oxalis* and graminoids in all of the plots, with a lack of overstorey shrubs, has been shown to be the general outcome of passive restoration methods in sand fynbos (Holmes, 2008). It has been found that short-lived species such as these have the most persistent propagules under *Acacia* invasion (Holmes, 2002). The presence of these non-invasive pioneer species could also be positive as these plants provide important ground cover where other fynbos species are absent, which helps counteract soil erosion (Holmes et al., 2000). Species from the Proteaceae and Ericaceae, which are dominant families in pristine CFSF, were completely absent from the plots in this study. This supports suggestions that fynbos restoration requires the input of propagules for recovery of these structurally important growth forms (Holmes and Richardson, 1999).

High variability of indigenous species composition found in the control plots is likely related to variation in the regenerating *A. saligna* stand, as well as to differential responses by the regenerating fynbos species. Indigenous fynbos species vary in their sensitivity to *Acacia* invasion (Holmes and Cowling, 1997), indicating that some of the germinating plants may be more sensitive to competing with *A. saligna* than others. It is also likely that some sensitive species were able to survive in plots that contained less dense *Acacia* regrowth, as the density of *A. saligna* regeneration varied across the study site. The relatively similar species composition following the two herbicide treatments, 'cut and poison' and 'foliar spray', is possibly a representation of more herbicide-resistant plant species. The species composition from these two treatments did not completely overlap, possibly because *A. saligna* saplings were not completely removed from the 'foliar spray' plots with resultant leaf litter impacting germinating fynbos plants (see Le Maître et al., 2011).

The plant assemblages recorded after the 'cut rootstock' treatment were unique, more species-rich and much less variable than those recorded after the other treatments. The contrasts in the plant assemblages between the different removal treatments validates the suggestion that the removal treatment utilized is likely to have a strong influence on the trajectory of vegetation recovery (Le Maître et al., 2011). Without comparison to a CFSF reference site, however, we do not know which follow-up treatment and restoration trajectory is more true to the composition of pristine CFSF.

4.4. Cost-effectiveness of treatments

Due to an inability to calibrate herbicide application equipment, the quantity applied to the stumps and plants in the two herbicide treatments was not measured. This not only becomes problematic in determining the true removal effectiveness and/or impacts of the 'cut and poison' and 'foliar spray' treatments, but it also limits the ability to determine the most cost-effective post-burn removal method. According to the treatment guidelines for *A. saligna* followed by the Working for Water (WFW) invasive species removal program, the recommended herbicide application for foliar spraying of a mature stand is 300 diluted liters (2.10 L of pure herbicide) per hectare (Bold, 2007). Due to the high density and cover of the post-burn regenerating *A. saligna*, the amount of herbicide required could potentially be greater and therefore more costly. The 'cut rootstock' and 'cut and poison' treatments were not only much more time consuming, but were extremely labor intensive, resulting in personal injury (in this case, a repeated strain bone fracture). Because the 'cut and poison' treatment required high labor input, herbicide, and potential harm to the workers, this is evidently the most costly treatment. Given this, and according to the WFW guidelines, we conclude that the 'foliar spray' removal treatment is the most cost-effective for removal of dense post-burn *Acacia*. The 'cut root stock' treatment times were, however, highly variable and this may still be a viable removal option in less dense re-growth.

4.5. Management implications

The indigenous species diversity and composition, as well as density of the regenerating *A. saligna*, was quite variable across the study site. This suggests that an adaptive management approach may be best in order to choose the most appropriate post-burn *A. saligna* removal treatment (Pat Holmes, personal communication), as the best treatment will also vary with these parameters. Where areas are large, with low recovery potential and high density *A. saligna* re-growth, using the 'cut root stock' treatment may be costly and time consuming. Post-burn *A. saligna* removal is a race against time as recruitment quickly results in closure of the canopy, to the detriment of any surviving indigenous species. In these instances, large-scale foliar spraying is probably the best method as it focuses on controlling *A. saligna*. Implementation of foliar spraying will need to be done using better equipment than used in this study to cope with the large scale and calibration that is needed to apply consistent quantities of herbicide.

As CFSF is a critically endangered vegetation type and an important goal here was restoring biodiversity, the 'cut rootstock' removal treatment should be strategically implemented. Areas with higher restoration potential and indigenous plant diversity should be delineated and this herbicide free treatment can be used to allow for the persistence of rare and sensitive species and assemblages. Although useful for the removal of mature *Acacia* stands, the 'cut and poison' method would not work in this instance because it is neither time- nor cost-effective.

Regardless of the post-burn follow-up treatment implemented, continued follow up and monitoring are necessary, not least because remnant *A. saligna* seeds may remain in the soil, and individuals can be missed in the removal process. Another reason for follow-up treatment and monitoring is that areas which are highly disturbed with low-diversity are vulnerable to re-invasion (Holmes et al., 2000). This makes it necessary to implement controls for secondary invasion, as is evidenced by this study where there was a proportionally high percentage cover of non-woody alien species in all treatment plots. Parallel with monitoring, promotion of CFSF pioneer species, such as those that currently dominate indigenous plant assemblages in this study, may be important as they help to protect the soil and provide microhabitats (Holmes et al., 2000; Holmes, 2008). In addition to continued control, there is a need for active restoration input. The fynbos seed bank is reduced under long-term invasion; functionally and structurally important species need to be re-introduced into the system (Holmes and Richardson, 1999; Holmes, 2002, 2008; Montoya et al., 2012). Although burning would have helped to reduce the excess soil nitrogen, the altered soils may still hinder the success of species from the Proteaceae and Ericaceae families (Holmes and Cowling, 1997).

5. Conclusion

Invasive *Acacia* species persist throughout the GCFR, threatening both species and ecosystems (Wilson et al., 2014). This study indicates that post-burn follow-up treatment can drastically reduce the density of *Acacia* regeneration and that the herbicide free 'cut rootstock' removal method is the most effective for restoring indigenous plant communities. The best method of removal, however, depends on the goals of each project and the density of the *Acacia* invasion. Although important, removal represents one step in the process of eradication and ecosystem restoration. Of particular concern is the damage to fynbos recovery caused by herbicides and the need to find practical alternatives for *Acacia* follow-up control in protected areas. For any eradication and/or restoration project to be successful, it is necessary that long-term control, monitoring, and restoration input are included in both planning and implementation. Thus, continued research and coordination with management decisions is needed, both in the Cape Flats Sand Fynbos and elsewhere in the GCFR.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at http://dx.doi.org/10.1016/j.sajb.2016.04.004.

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