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RESEARCH ARTICLE

Using fire, planting, and topsoil translocation for restoration in South African Fynbos: assessing the success and cost-effectiveness of various restoration treatments

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The successful restoration of endangered habitat types at a low cost is of vital importance. We tested how successful and cost-effective six combinations of restoration treatments were in the Cape Floristic Region of South Africa. Treatments were applied to an area that was primarily covered invasive alien grasses. Treatments that tested combinations of burning, direct sowing of seed, and planting of seedlings, aimed to test the synergistic effect of three treatment factors. Topsoil translocation aimed to determine if this, combined with planting, can restore native shrub cover and whether the addition of mulch and seed can improve overall success. The mechanical removal of invasive grasses, sowing and planting, were tested in the absence of fire. Both topsoil translocation treatments and the treatment that had been burned, sown and planted ("Burn-sow-plant") led to plant species richness resembling near-pristine plots and approximately 30% higher than the burn-only treatment ("Burn"). Native shrub cover was the highest for treatments which included planting. Survival for planted species was 36–41% higher for Burn-sow-plant. The contribution of sowing was low, with 9–16% of species sowed being present across treatments. We show that topsoil translocation can be successful if combined with planting. Burn-sow-plant was most successful at a reasonable cost, despite planting being expensive, whereas Burn and Burn-sow were less successful, but cheaper. The treatment in which grass was removed before sowing and planting was the least successful and most expensive. We conclude that the most successful treatments are not necessarily the most expensive.

Key words: active restoration, Cape Flats Sand Fynbos, Cape Floristic Region, Cape Town, ecological restoration, invasive alien grass, planting, sowing

Implications for Practice

- Planting plants increases restoration success in shrublands, and despite costs, should be considered, even if just in part of the restoration area from which these can act as nuclei.
- Although the application of fire and topsoil translocation from near-pristine sites respectively aids in reducing competition from alien species, the removal of invasive alien plants should continue in small restoration sites with a history of invasion.
- More research is needed on the specific germination requirements of seed, as this might yield more success with sowing as a treatment to reintroduce threatened plants.
- Topsoil translocation should only be used as a last resort and not as a mitigating solution for development, as the translocated system will simply resemble the lost one.

Introduction

In fire-driven ecosystems such as Mediterranean systems, prescribed burning after alien clearing can stimulate the germination of soil-stored native seed banks (Vlok & Yeaton 1999; Montenegro et al. 2004; Keeley 2012). Fire is a cost-effective treatment for the restoration of fire-prone systems and far less labor-intensive than other methods (Jackson et al. 1982; Musil et al. 2005). Globally, the use of topsoil translocation for ecological restoration has increased in recent years as an alternative to burning (Fowler et al. 2015). Few studies have used this in Mediterranean systems other than south-eastern France (Jaunatre et al. 2014; Bulot et al. 2017; but see Holmes 2001). Topsoil

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that is translocated from intact donor sites to degraded areas not only aids in translocating seeds, but also potentially translocate other plant propagules such as root fragments, rhizomes, corms, tubers (Ferreira et al. 2015) as well as microorganisms (Buisson et al. 2018) that are essential for germination and growth. However, the outcome of topsoil translocation is largely dependent on a rapid operation, with minimal or no stockpiling (Koch 2007*a*), at the right time of the year (usually autumn for Mediterranean areas) (Bulot et al. 2014), as well as translocating topsoil to an area the same size as the donor site (1:1) (Buisson et al. 2018). Topsoil translocation has been shown to be successful if combined with other methods that are usually used with burning, such as sowing and planting, although the woody shrub component of vegetation may remain difficult to restore with this method (Pilon et al. 2019).

Native seed forms an important component of restoration projects (Nevill et al. 2018) because it can facilitate and accelerate the recovery of vegetation (Palma & Laurance 2015). However, when soil seed banks are small, relying on sowing can pose a challenge as seed harvesting is time-consuming (Broadhurst et al. 2016), while commercial native seed stock is not always available and can be costly (Cross et al. 2020).

Because most plant species growing from seed banks have been found to be annuals and short-lived forbs and shrubs (Holmes & Richardson 1999), seed of perennial shrub species in Mediterranean systems may not survive for long in the seed bank (Holmes 2008). Therefore, it is beneficial to also plant propagated plants after a prescribed burn (Hall et al. 2021). The planting of propagated plants in burnt areas is uncommon in South Africa (Ruwanza et al. 2013), while globally, there is an insufficient number of studies to conclusively show benefits of these methods (Palma & Laurance 2015; Nolan et al. 2021). Furthermore, propagation of plants can be costly (Ruwanza et al. 2013), especially for large areas (Rodrigues et al. 2009). This highlights the trade-off between success versus cost, as propagated plants often survive better (Cole et al. 2011)—which increases restoration success but, in turn, costs more. Determining the success of fire in combination with sowing and planting could potentially assist to identify the most cost-effective combination of treatments.

The question whether the effect of mulch—after soil translocation—enhances seed germination and seedling establishment, has received limited attention in Mediterranean systems (Holmes 2001; Tormo et al. 2007; Benigno et al. 2013). The high costs associated with harvesting and applying mulch have been found to be unjustified in terms of the little benefit it provided in Australian *Banksia* woodlands (Rokich et al. 2002)—although, this may be different for other Mediterranean systems.

Cost-effectiveness of different restoration treatments is important, but often ignored (Acosta et al. 2018). Ecological restoration is expensive; therefore, it is imperative that research considers not only restoration treatments that are potentially most successful, but also those that are most cost-effective. Even more so, research should provide guidance on the interaction between success and costs. Success can be measured in various ways, but often vegetation structure, species diversity, and abundance suffice (Wortley et al. 2013). While it is not necessary to reintroduce all species to reinstate ecosystem functions, the higher the plant species richness, the more resilient the ecosystem will be in the long term (Alberti & Marzluff 2004).

Our study took place within the Fynbos ecosystem of the Cape Floristic Region (CFR) in South Africa. The CFR is characterized by a Mediterranean climate, with winter rainfall and hot, dry summers. Fynbos is an evergreen shrubland of which more than 5% of plant communities usually consist of Restionaceae. Furthermore, Ericaceae and Proteaceae shrubs usually dominate the landscape, with ephemerals and geophytes being mainly present after the occurrence of a fire (Rebelo et al. 2006). The majority of woody shrubs within Fynbos survive as obligate reseeders while the seed banks of Restionaceae and Proteaceae are mostly long-term persistent (Holmes & Newton 2004). Similar to other Mediterranean systems globally, the survival of Fynbos is strongly influenced by fire (Keeley 2012; Geerts 2021). If a native seed bank no longer exists, restoration attempts should also include the reintroduction of native plant species, either through sowing or planting (Pretorius et al. 2008). Cape Flats Sand Fynbos (CFSF), a lowland Fynbos type, is prone to disturbance, especially through the invasion of alien annual grasses (Rebelo et al. 2006).

Here, we aim to test the synergistic effect of different combinations of restoration treatments in a small urban nature reserve (i.e. fire, sowing of annual and perennial forbs, shrubs and geophytes, planting of woody shrubs, topsoil translocation, mulch, and the removal of invasive grass layers). Specifically, we addressed the following questions: (1) Does the synergy of a prescribed fire, sowing and planting provide greater native plant species richness than a prescribed fire on its own? (2) Can topsoil translocation and subsequent planting alone lead to greater plant community recovery than using these factors with the addition of mulch and seed? (3) Does the planting of woody shrubs, as an active restoration treatment used in combination with either a prescribed fire or topsoil translocation, restore native shrub cover? And (4) is the use of a prescribed fire in combination with sowing and planting native plants more cost-effective than topsoil translocation in combination with planting?

Methods

Study Area

Our study area was located within Milnerton Racecourse reserve, a small (22 ha) urban nature reserve in the CFR of South Africa (Fig. 1). Milnerton Racecourse reserve is one of the last remaining remnants of CFSF, a critically endangered vegetation type found within the CFR (Rebelo et al. 2006).

In 2015, the nature reserve acquired 3 ha of degraded land adjacent to the existing reserve (Fig. 1; Sites 1 & 2). The site

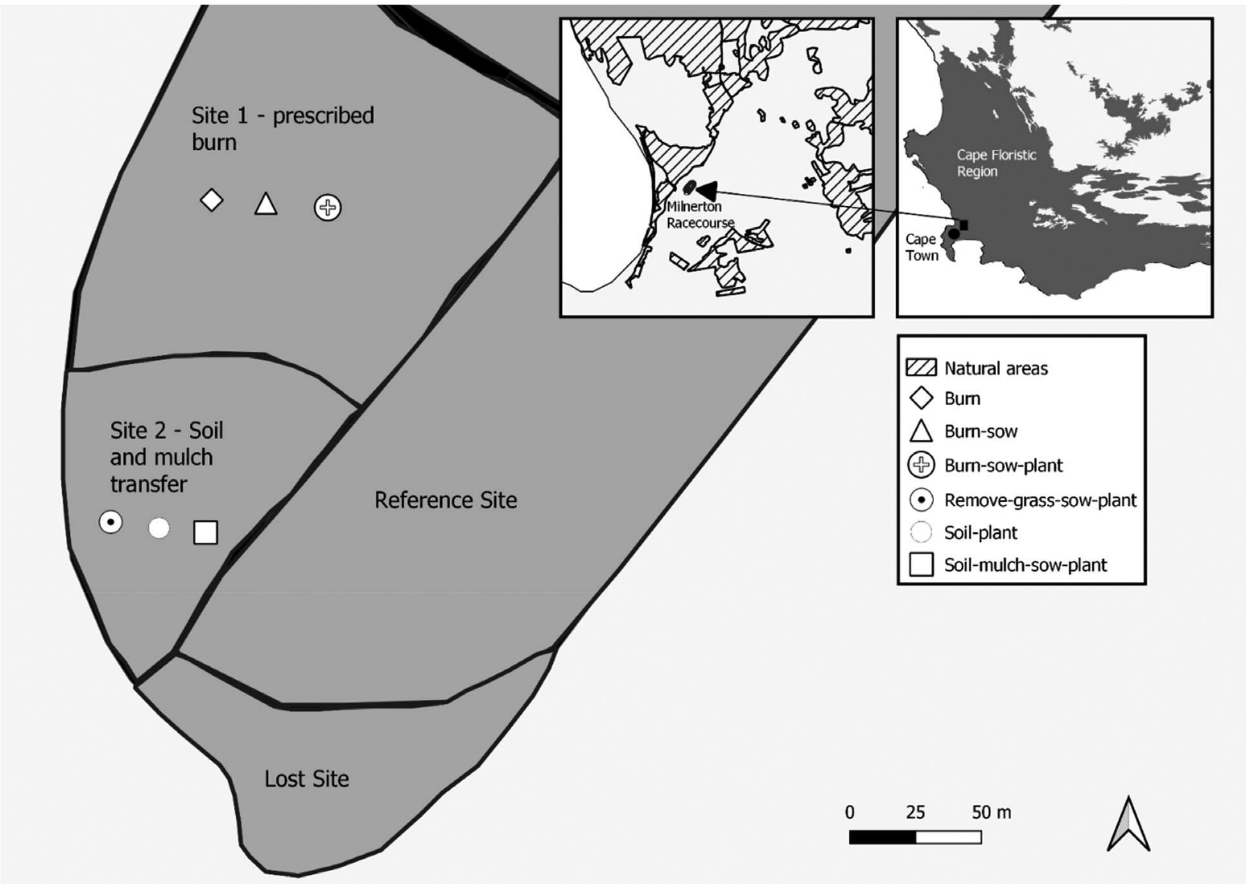

Figure 1. The study was located within Milnerton Racecourse, a small urban nature reserve in the Cape Floristic Region (CFR), South Africa. The site was divided into four areas as indicated on the map—the lost site, the reference site as well as two sites (Sites 1 and 2) that each received a number of different restoration treatments. Right inset: The extent of the CFR as well as the location of Milnerton Racecourse. Left inset: The remaining natural areas around the study site.

was taken over in 2017 by the City of Cape Town, to be restored and managed as a nature reserve. As part of a land exchange agreement, 0.561 ha of well-managed CFSF was lost (Fig. 1; lost site). The topsoil and resultant mulch from the vegetation on this site that was lost were translocated to the degraded area (Fig. 1; Site 2). Control plots within the reference site were selected in the adjacent intact nature reserve, within part of the same vegetation community of the site that was lost. The reference site as well as lost site where topsoil and mulch were sourced from, were in near-pristine condition. Areas which received the Burn, Burn-sow, and Burn-sow-plant treatments were previously used as a polo field and had an estimated 40% cover of alien grass species (Table 1). The Soil-mulch-sow-plant and Soil–plant treatments were applied to an area that was previously used for horse-training activities (Table 1). The area consisted of several parade rings as well as an old road covered with nutrient-rich laterite (Fig. 2A & 2B). The Remove-grass-sow-plant treatment was applied to the areas next to the horse parade

Table 1. A summary of the factors included in each restoration treatment. Individual treatment components included the translocation of topsoil and mulch, the mechanical removal of an invasive grass layer, the application of fire, the sowing of seed as well as planting.

Site		*Burn*	*Sow*	*Plant*	*Soil*	*Mulch*	*Remove grass*
1	Treatment 1	✓					
	Treatment 2	✓	✓				
	Treatment 3	✓	✓	✓			
2	Treatment 4			✓	✓		
	Treatment 5		✓	✓	✓	✓	
	Treatment 6		✓	✓			✓

Figure 2. Vegetation recovery 3 years after restoration. An aerial image of Site 2, 1 year prior to treatment application (A) is compared to an image of the area after topsoil translocation and mulch application to roads and parade rings (B). The lost site (C) is compared to the site where topsoil was translocated to (F). The initial condition of Site 1, where fire was applied (D) is compared to the condition of the site after treatment (E). Source: Photos L. Retief.

rings within Site 2 (Table 1; Fig. 1). An estimated 60% of this site was covered with alien grass species, with a low abundance of other invasive weed species present. Control measures were put into place to exclude herbivores (Cape grysbok [*Raphicerus melanotis*]) from the restoration site. Raptor perches were set up to limit possible granivory. It must be acknowledged that Cape Town experienced a severe drought during the 3 years prior to restoration treatments taking place with an annual rainfall which were at 20–47% lower than the normal average.

Restoration Treatments

Topsoil and Mulch Translocation. The translocation of topsoil from the lost site (Fig. 2C) to a part of Site 2 (Fig. 1) took place during autumn 2017 before the first winter rains. The lost site received a prescribed burn, 7 years prior to topsoil translocation, which would have allowed enough time for the perennial species present to form seed. The vegetation was cut to ground level with brush cutter machinery and passed through a shredder. The resulting mulch was mixed with topsoil to create a soil–mulch blend. This was immediately spread out on Site 2 with minimal stockpiling taking place. Laterite that was historically brought in to cover roads in Site 2 was removed from a depth of 200 to 300 mm and replaced with either topsoil or a soil–mulch blend of the same depth from the lost site. Topsoil was translocated at a ratio of 1:1 (Buisson et al. 2018).

Prescribed Burn. A prescribed burn was done on Site 1 (Figs. 1 & 2D). This took place during autumn on a mild sunny and windless day. A head-fire was used, together with drip torches and leaf blowers.

Reintroduction of Plant Species. Seeds and cuttings were sourced from the Milnerton Racecourse nature reserve. Seeds were treated with a dry smoke method. Sowing was done by broadcasting and embedding seeds, 1 day after the prescribed burn. Seeds of annual and perennial forbs, shrubs, and geophytes were sown at 10 kg/ha within 24 monitoring plots (Esler et al. 2014). The seeds of 18 different plant species were sown (Table S1). For propagation, plant material was propagated into a mix of 50% sand from the restoration site and 50% nursery soil. Plants were hardened off within 1-kg bags, in full sun, 2 weeks prior to planting. Planting in the restoration site took place after a cold front that had occurred during winter. Perennial shrubs were planted out at 2800 plants/ha within 24 monitoring plots. Nine different plant species were planted out (Table S1). Plants were planted in groups or "islands" of three to four plants to ensure better cover and micro-climates for the germination and survival of seeds (Cowell 2014) as well as to allow planted plants to facilitate more protection against extreme environmental conditions such as wind and heat (Padilla & Pugnaire 2006).

Alien Grass Removal. The plots used for the Remove-grass-sow-plant treatment (Table 1) were primarily covered in alien grass species. Within these plots, the grass layers were removed together with the top layer of soil (10 mm). Removal was done manually with spades and material was removed from the site.

Sampling Design and Vegetation Monitoring

Data were collected once a year, during spring (September and October). The existing roads and parade rings in the restoration site, where there was no vegetation cover, received topsoil and mulch translocation treatments as an attempt to relocate the lost system to these bare areas in the study site. Due to the limited area that was available for topsoil and mulch translocation as well as the limitations of a small urban study site with the risk of edge effects compromising plots, not all restoration treatment factors could be tested separately. Resource limitations also led to seed stock for sowing being too low and therefore topsoil translocation was tested without sowing. Fire was applied to an area separate from where translocated topsoil and mulch were spread, due to the burnt area having sufficient vegetation matter to fuel a fire. This was to ensure a quick and effortless prescribed burn in a highly urbanized area. Data were collected prior to the restoration treatments taking place, from six single rectangular 50 m^2 plots within the area that was lost. The distance between plots in the site that was lost was approximately 90 m. Thereafter data were collected for 2 more years after restoration treatments had taken place. During these 2 years, data were collected from 42 single rectangular 50 m^2 plots. Six 50 m^2 replicas were used in separate areas for each of the six restoration treatments within an experimental site, as well as for a reference site, which was approximately 50 m from the experimental site within the neighboring nature reserve (Fig. 1). The distance between plots within the reference and experimental site ranged from 89 to 128 m. Year 1 and year 2's data collection took place respectively, 5 and 18 months after restoration treatments. As it takes time for vegetation parameters such as cover and density to be representative of an intact site, species richness was recorded, as this is a short-term ecological metric that can be used to evaluate the effectiveness of interventions (Suding 2011). Species were classified according to status (native and alien), and growth form (graminoid, forb, and shrub). The presence of plant species that were either sown or planted was recorded. As a community attribute, proportionate cover was recorded as growth form (graminoid, forb, shrub, and sand).

Financial Analysis

An analysis of the costs of restoration treatments was done. Costs were calculated in South African Rands/ha (1 ZAR = 0.069 USD; 12 July, 2021). All expenses of each restoration factor were calculated, including labor, as well as the resources needed for the preparation and execution of treatments. Where it was cheaper and practical, the hiring costs of items were used as operational costs to avoid expensive capital expenditure. Two quotations from either a business or service provider were obtained for items where applicable, from which the average was then calculated. In terms of the prescribed burn, one quotation was obtained from an external service provider. This was compared and averaged with the estimated cost of sourcing all resources and conducting a burn internally by nature reserve management. The wages of junior/labor staff were calculated according to the South African National Minimum Wage of the Expanded Public Works Programme for 2021 (South Africa 2021). Wages of midlevel staff were calculated according to the average South African salary for 2021 (Business Tech 2021). Wages per day were calculated according to 8-hour shifts. All total costs were scaled to depict the cost of each treatment per 1–50 ha (Blignaut et al. 2014). The cost of treatments was compared with the success of treatments in terms of native species recovery.

As the cost of many of the resources used for fire as a restoration treatment remains the same even as the number of hectares increases, this treatment was scaled as follows: 20 crew members with tools were allocated for a 1 ha fire, which was increased with one crew member with tools for every 5 ha added; three fire fighting utility trucks were assigned for a 1 ha fire, which was increased with one fire fighting utility truck for every 10 ha added; and 60 L of drip torch fuel were assigned to a 1 ha fire, which was increased with 60 L for every 10 ha added.

Statistical Analyses

Statistical analyses were conducted in R version 4.0.3 (R Core Team 2020) and generalized linear mixed models (GLMMs) were fitted using the *glmmTMB* package (Magnusson et al. 2020). Using model selection, we determined the effect of the following: (1) restoration treatment, years after treatment, and their interaction on native species richness (Poisson error distribution and log link function); (2) restoration treatment, years after treatment, and their interaction on alien species richness (Poisson error distribution and log link function); (3) restoration treatment, growth form, and their interaction on native species richness (Poisson error distribution and log link function); (4) restoration treatment, years after treatment, and their interaction on planted species survival (beta error distribution and logit link function); (5) restoration treatment, years after treatment, and their interaction on sowed species survival (Beta error distribution and logit link function); (6) restoration treatment, years after treatment, and their interaction on native species cover (beta error distribution and logit link function); (7) growth form, years after treatment, and their interaction on species cover (beta error distribution and logit link function).

"Plot" was included in each model as a random effect to account for variability between plots and to quantify variability not only temporally but also spatially as well. Different models were compared via dredging, an automated procedure implemented by the *MuMIn* package (Barton 2018), to identify variables that best explain the variability in each model. Models were compared using information theoretic model procedures based on Akaike's information criterion (AIC; Burnham et al. 2011). The best-fitting model was chosen using the second order AIC value, AICc. The model with the lowest AICc value was chosen as the best-fitting model as it had the smallest information loss, even if it did not include all the explanatory variables. For each response variable, only one model remained after model selection. Results of these best-fitting models are reported here. Significant mean differences were separated with Tukey's honest significant difference test using the *emmeans* package (Lenth 2018).

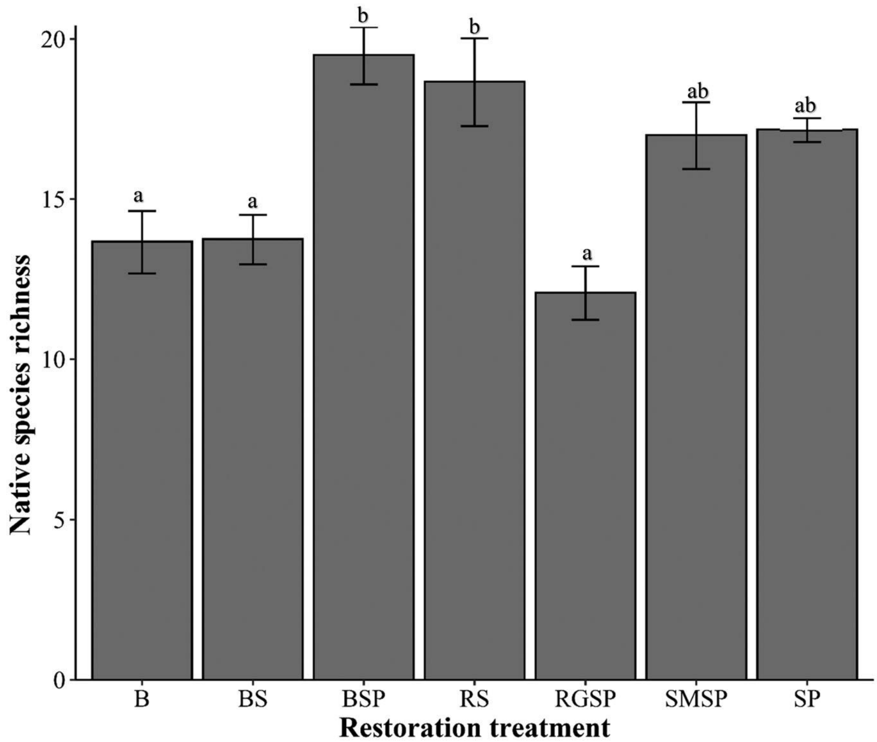

Figure 3. The effect of restoration treatment on native species richness (generalized linear mixed models). Error bars indicate ± SE. The letters above the error bars indicate significant differences between treatments where $p < 0.05$. B, Burn; BS, Burn-sow; BSP, Burn-sow-plant; RGSP, Remove-grass-sow-plant; RS, reference site; SMSP, Soil-mulch-sow-plant; SP, Soil–plant.

Results

Effects of Restoration on Vegetation Parameters

In an analysis of the effect of restoration treatment, years after treatment and their two-way interaction on native species richness, "year" was not significant in the best-fitting model. For Burn-sow-plant (Fig. 2E), Soil-mulch-sow-plant, and Soil–plant (Fig. 2F), native species richness was similar to the reference site ($p = 0.99$; Table S2; Fig. 3). Burn, Burn-sow, and Remove-grass-sow-plant had a significantly lower native species richness than the reference site ($p < 0.01$; Table S2; Fig. 3).

There was a significant interaction in the effect of restoration treatment and year on alien species richness ($p < 0.01$; Table S3). There were significantly less alien species in the second year ($M = 5.6 \pm 0.3$ [SE]) compared to the first year ($M = 7.2 \pm 0.3$ [SE]) ($p < 0.01$; Tables S3 & S4). Of all treatments, the lowest number of alien plant species was found within Burn-sow, during the second year ($M = 4.5 \pm 0.7$ [SE]; Table S4). The three restoration treatments that involved the use of fire all had a lower number of alien plant species ($M = 5.5 \pm 0.2$ [SE]) than treatments that did not involve the use of fire ($M = 7 \pm 0.4$ [SE]; Table S4).

The treatments that included the use of planting as a restoration factor, all had a significantly higher native shrub cover than those treatments that did not, including the reference site ($p < 0.05$; Table S5; Fig. 4). Native forb cover was significantly lower than the reference site within grass-removed-sow-plant ($p < 0.01$; Table S5; Fig. 4).

Of the plant species that were planted, more than 96% of species were still present within all treatments after 1 year. There was a significant reduction in the presence of planted species by the second year for all treatments ($p < 0.001$; Table S6; Fig. 5A), except for Burn-sow-plant with an 81% survival. The overall presence of sowed species was low, ranging from 9 to 16% for all treatments during the second year. During the first and second year, the presence of sowed species was significantly higher within Soil-mulch-sow-plant (16–20%; $p = 0.01$; Table S7; Fig. 5B) than Remove-grass-sow-plant. Burn-sow-plant had the second highest presence of species sowed during the first year (19%) as well as the second year (12%).

In the analysis of restoration treatment, years after treatment, and their interaction on species cover "restoration treatment" was not included in the best-fitting model. There was a nonsignificant effect of "years after treatment" on species cover ($p = 0.967$; Fig. 6). There was a significant interaction between the effect of growth form and years after treatment ($p < 0.001$;

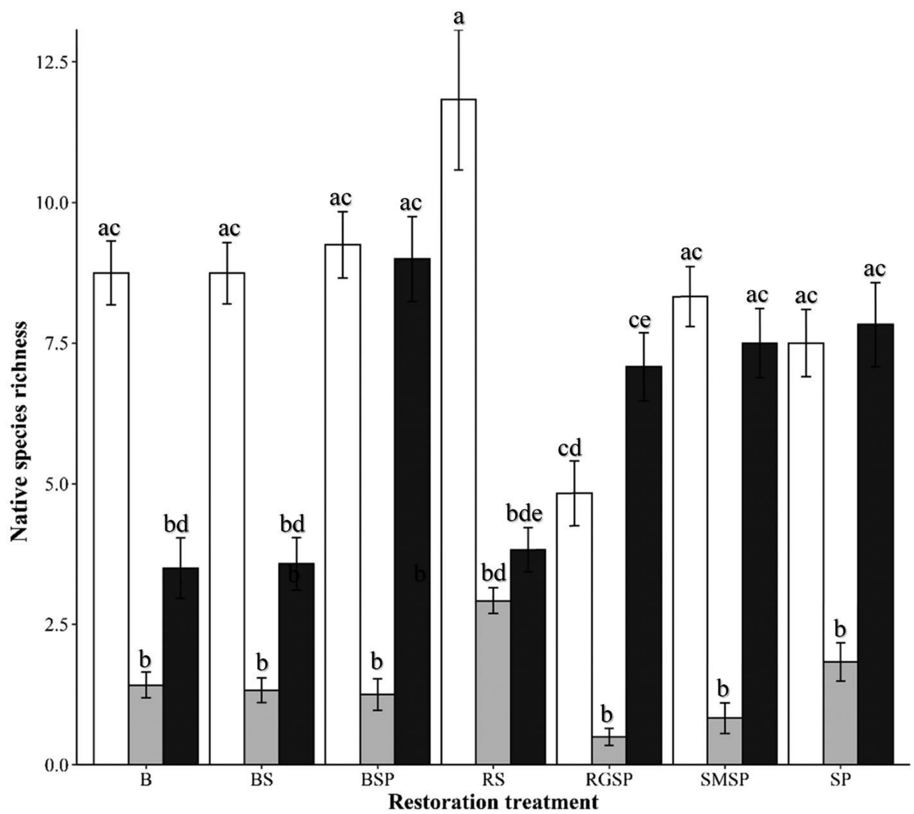

Figure 4. The effect of restoration treatment on native species richness per growth form (generalized linear mixed models). The growth forms were recorded as forbs, graminoids, and shrubs. Error bars indicate ± SE. The letters above the error bars indicate significant differences between treatments where $p < 0.05$. B, Burn; BS, Burn-sow; BSP, Burn-sow-plant; RGSP, Remove-grass-sow-plant; RS, reference site; SMSP, Soil-mulch-sow-plant; SP, Soil–plant.

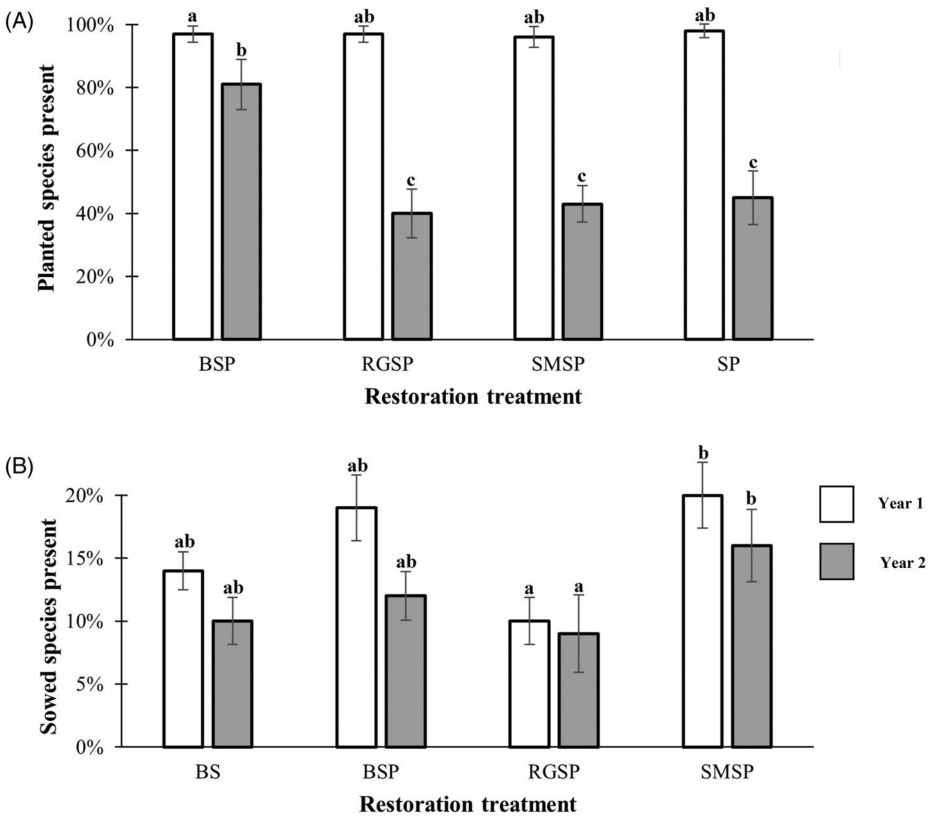

Figure 5. The percentage of planted (A) and sowed species (B) surviving in the different restoration treatments after 1 and 2 years. Error bars indicate ± SE. The letters above the error bars indicate significant differences between treatments where $p < 0.05$. B, Burn; BS, Burn-sow; BSP, Burn-sow-plant; RGSP, Remove-grass-sow-plant; SMSP, Soil-mulch-sow-plant; SP, Soil–plant.

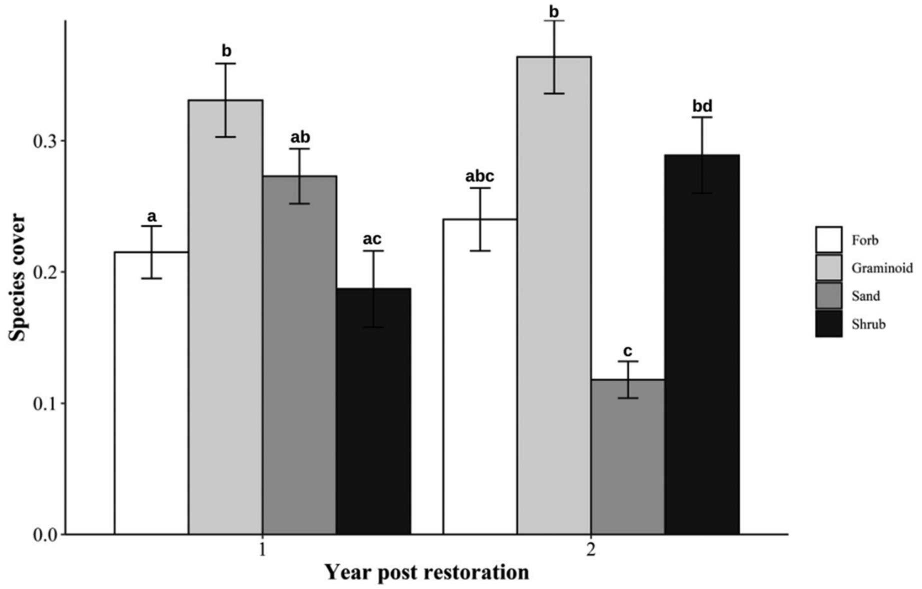

Figure 6. The effect of growth form, years after treatment, and their interaction on species cover. The cover of growth forms was recorded as forbs, graminoids, sand (bare ground) and shrubs after 1 and 2 years. Error bars indicate ± SE. The letters above the error bars indicate significant differences between treatments where $p < 0.05$.

Table 2. The combinations of restoration treatments used in this study and the cost/ha as well as cost/50 ha. Costs were calculated in South African Rands/ha (1 ZAR = 0.069 USD; 12 July, 2021).

Restoration treatment	*Cost/ha (ZAR)*	*Cost/50 ha (ZAR)*
Soil–plant	R153,594	R7,679,700
Remove-grass-sow-plant	R422,791	R21,139,600
Soil-mulch-sow-plant	R264,696	R13,234,850
Burn	R36,381	R55,951
Burn-sow	R123,173	R4,395,551
Burn-plant-sow	R234,172	R9,945,551

Fig. 6). There was a significant increase in shrub cover (10%), from the first to the second year (Fig. 6).

Financial Analysis

At R36,381/ha, the Burn treatment is the most cost-effective (Tables 2 & S8–S11). When the cost of this treatment was scaled to 50 ha, the resources used largely remained the same, with a minimal increase in cost (Table 2). The costs of all other treatments showed a linear increase when scaled from 1 to 50 ha. At R422,791/ha, Remove-grass-sow-plant was the most expensive (Table 2). Burn-sow-plant (R234,172/ha) and Soil–plant (R153,594/ha) were estimated to have relatively moderate costs (Tables 2, S12, & S13). Burn-sow-plant was identified as one of the most viable options, as it was most successful (highest native species richness) and had the fourth lowest cost (R234,172/ha) (Fig. 7). In terms of the treatments that involved the movement of soil and/or mulch, Soil–plant was the most successful treatment at a relatively moderate cost (R153,594/ha) (Tables S14 & S15; Fig. 7) Remove-grass-sow-plant had the least success and had the highest cost (R422,791/ha) (Table S16; Fig. 7).

Discussion

Our study took place in a small urban nature reserve, in a critically endangered Mediterranean vegetation type with limited extent. This, in addition to the small size of the area from which topsoil could be translocated and the differing site histories within the reserve, posed limitations to our experimental design. Nonetheless, we are able to show that fire combined with planting can restore native plant species richness, and specifically shrub cover, to that of a near-pristine reference site. We also show that the appropriate translocation of topsoil (with a soil seed bank) and subsequent planting can be successful in restoring Mediterranean plant communities such as CFSF.

Fire

The exposure to heat and smoke from fire plays an important role in breaking the dormancy of seed and stimulating germination in fire-prone Mediterranean systems (Keeley 2012; Hall et al. 2016; Zomer et al. 2022). Although too frequent fire intervals have demonstrated a loss of plant community diversity in Australian Mediterranean systems, (Fischer et al. 2009) the application of fire at the correct intervals can assist in removing invasive grass layers and exposing soil for successful sowing and planting in the absence of competition (Keeley 2006). In a restoration study done by Waller (2013), it was evident that the use of fire alone with no other restoration treatments kept native species richness relatively low. This is consistent with our study—not only for Burn, but also for Burn-sow.

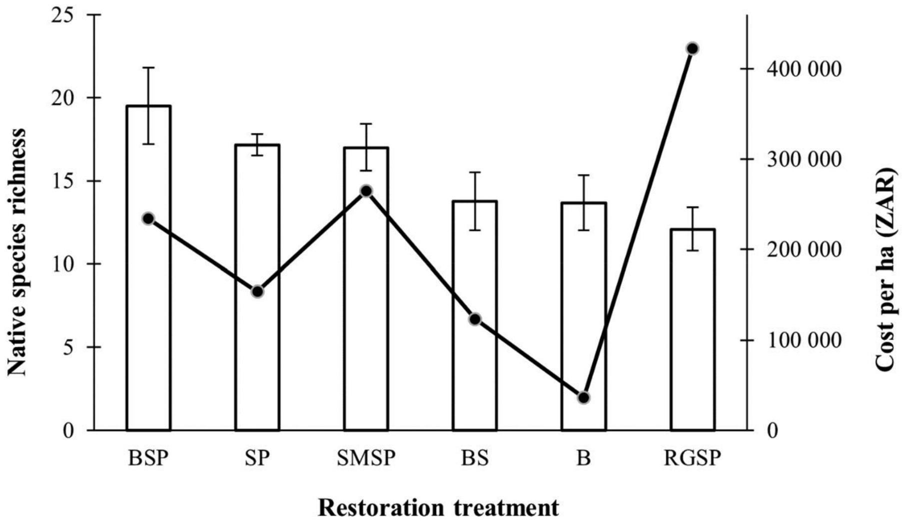

Figure 7. The success (indicated by bars) and costs (indicated by a line) of the six restoration treatments. Success is measured here as native species richness averaged across the 2 years post-restoration. The cost of each restoration treatment was calculated by including all labor and resources required. Errors bars indicate ± SE. Costs were calculated in South African Rands/ha (1 ZAR = 0.069 USD; 12 July, 2021). B, Burn; BS, Burn-sow; BSP, Burn-sow-plant; RGSP, Remove-grass-sow-plant; SMSP, Soil-mulch-sow-plant; SP, Soil–plant.

Sowing

Although the presence of sowed species was slightly higher when used in combination with fire and planting (Greet et al. 2020) as well as the application of mulch, sowing had no significant effect on restoration success. This seemingly low contribution of pre-treated seeds sowed after the fire is in contrast with the findings of Hall et al. (2016), whose results showed this to be a successful combination of restoration treatments. Our results correspond with a global review done by Palma and Laurance (2015) as well as those of Gaertner et al. (2012), where the application of seed had a low success rate or no significant effect on native species richness. However, these methods are typically applied following the clearing of woody alien invasive species that are stimulated by fire. In contrast, prior to restoration, woody alien plants species were absent from our study site, which resembled an old agricultural field with mostly alien grass species present.

Planting

The high survival rate of clustered groups of planted species in our study supports the notion that some plants benefit from growing in close proximity of each other by facilitating more protection against environmental factors such as wind and heat (Padilla & Pugnaire 2006). It has also been found that with sowing alone, it takes longer for larger shrub species to establish themselves (Holmes & Richardson 1999). Thus, the inclusion of planting can ensure a high native shrub cover (Pyke et al. 2020). Furthermore, case studies in Portugal and Australia have found that plant species with a higher resilience against invasion should be favored to be included in restoration projects (Fischer et al. 2009; Duarte et al. 2020). Competition from alien invasive plant species is known to be a major constraint to revegetation (Greet et al. 2020) and therefore fire can be used to reduce competition posed to newly planted plants (Holmes 2008). Studies in the Chilean matorral have demonstrated that native species often return in higher numbers in areas that have been burnt as opposed to those that have not (Holmgren et al. 2000). Fire also acts as an important mineralizing agent offering a post-fire flush of nutrients made available (Dannenmann et al. 2018; Xofis et al. 2023). This may explain the higher survival rate of planted species in the burnt treatment.

Low Germination of Sowed Species

Sowing within mulched areas can assist germination of seed by providing shelter and moisture (Rokich et al. 2002; Benigno et al. 2013), which corresponds with our results. However, we did not test the factors that could have affected the overall low germination of sowed species. These factors could have included a lack of germination cues or other environmental factors such as the prevailing weather conditions. Furthermore, it could have been due to direct competition with alien grass species, as previously observed in western Australia as well as South Africa (Standish et al. 2007; Nsikani et al. 2020). The leachate of *Avena fatua* (which was abundant in the study site) is known to have an allelopathic effect on the germination of seeds (Tinnin & Muller 1972).

The Mechanical Removal of Invasive Grasses

The low success rate of the Remove-grass-sow-plant treatment could possibly be due to the high remaining grass component within the surrounding habitat of treatment plots, that were still distributing seed into the plot area. Alien grass species grow faster than native plants (Acosta et al. 2006; Ruwanza 2017) and can produce high quantities of seed. There is a possibility that some of the existing native seeds could have been removed with the invasive grass layers or that the results here are linked to the features of the tested site itself, which had a very distinct history of alien grass species being present. The seed banks in long-invaded areas have been found to be less resilient which may lead to a higher risk of secondary invasion (Marchante et al. 2011).

Topsoil Translocation

The success of topsoil translocation can depend on the soil containing an adequate seed bank (de Villiers et al. 2004; Buisson et al. 2018) as well as other plant propagules (Ferreira et al. 2015) in the soil, which is why it is recommended that topsoil application must be used with other restoration techniques (Fowler et al. 2015; Pilon et al. 2019). Studies have shown that topsoil translocation can be successful in suppressing a seed bank of invasive weeds and grass species (Bulot et al. 2014; Ruwanza 2020). Due to topsoil having been translocated from a near-pristine site, we could test the method in the absence of a large seed bank of either alien grass or weed species, which reduced overall competition with native species (Marchante et al. 2011). Similar to studies done in the northwestern Mediterranean grasslands of Europe (Buisson et al. 2021), the translocated topsoil could provide optimum conditions for the successful rooting of plants due to soil structure and soil processes having been in place. Using a combination of planting with topsoil translocation, this successfully facilitated the restoration of a native plant community, including native shrub cover, and a system that resembled the area that was lost. Holmes (2001) found a high native species richness within areas that received topsoil application with sowing, even though certain plant species which are important key structural elements in the vegetation type were still missing. This could potentially be addressed by incorporating the planting of selected plant species as a restoration factor. The treatment Soil–plant was the only treatment where *Muraltia satureioides* was present, a species that was found in the lost site and was moved as seed within the translocated topsoil. This species is part of the Polygalaceae family, a taxon known to be problematic to cultivate (Nichols 2005). Topsoil translocation does not constitute adequate mitigation for the loss of endangered habitat (Bullock 1998), nor does it replace in situ conservation (Jaunatre et al. 2014) and often has no net gain for conservation. However, it could be seen as a means to relocate and introduce plant species that are difficult to cultivate from either seed or vegetative cuttings.

Our results only represent data collected for 2 years after restoration treatments, which is relatively short term for restoration studies in Fynbos (Hall et al. 2021) and Mediterranean systems in general (Vallejo et al. 2012). The sowing of native shrub species has been shown to be unsuccessful if done during years of drought in Californian shrublands (Cione et al. 2002) as well as woodlands in western Australia (Grose 2013). The lack of success with some of our restoration treatments tested may have been due to one of the worst droughts in recent history that was experienced in the region during the 3 years prior to restoration treatments taking place (Padilla et al. 2009; Archer et al. 2019) or competition with alien grasses (Fagúndez & Lema 2019). It is unlikely that the low success rate of some treatments was due to herbivory or granivory, as antelope were excluded from our study area by a fence and no rodent activity or excessive activity of other granivores, such as ants, was noted.

The Cost of Treatments

Similar to Waller (2013), we found that the number of factors included in our restoration treatments corresponded positively with the cost, but the most successful treatment was not necessarily the most expensive. We carried out restoration treatments with methods that were suited to a small-scale urban restoration project. In larger areas, mechanized options could be used which could be more cost-effective (Holmes & Richardson 1999; Campos-Filho et al. 2013; Navarro-Pedreño et al. 2017). Despite this, the relative difference in cost between all the tested restoration treatments should remain constant, even at other restoration sites. Burn-sow-plant proved to be the most successful treatment, but also more expensive relative to Burn and Burn-sow. It is widely known that the use of seed in restoration projects is cheaper than planting (Ruwanza et al. 2013; Greet et al. 2020; Holmes et al. 2020; Hall et al. 2021), but we show here that the added success achieved by including planting, allows restoration practitioners to weigh costs against success.

Burn was the only restoration treatment that did not show a linear increase in terms of cost. The concept of economy of scale became relevant since, as the size of the burn area is increased to a scale larger than 50 ha, the cost per hectare comes down (Jackson et al. 1982). Soil–plant was far less expensive and more successful than Soil-mulch-sow-plant, regardless of the addition of mulch and seed to the latter, which also did not contribute to restoration success. The high cost of the Soil-mulch-sow-plant treatment was primarily due to the labor necessary for mulch application, seed collection, and sowing. In contrast, topsoil application can be applied in a shorter time with less labor needed (Koch 2007*b*). This is a cost-effective restoration treatment which is suited to Mediterranean systems with high levels of biodiversity, where technical knowledge of germination treatments and propagation are often limited (Fowler et al. 2015). However, we must acknowledge that in the context of our study, topsoil transfer creates a situation where there is no net gain in terms of biodiversity. This method is thus only applicable to specific situations, where the integrity of small conservation sites is compromised or where proposed land development is unavoidable. The viability of topsoil translocation as part of large-scale restoration projects as well as a restoration factor tested without other interventions such as planting will have to be explored. The most expensive and least successful treatment was Remove-grass-sow-plant. The method was time-consuming and ultimately amounted to high costs. These high costs associated with the method of removal with spades corresponded with that of mechanical methods used by Steers and Allen (2010) in Californian shrubland. In said study, it was found that the disturbance of mechanical removal in the absence of fire may facilitate the reestablishment of invasive grass species to an extent worse than prior. The challenging removal of these grass species and the high costs associated with it, emphasize the importance of the use of fire for the restoration of Mediterranean systems to reduce the competition posed by invasive alien grass (Keeley 2006).

In conclusion, we recommend that, with limited funds, the application of fire is used. Where an extensive budget is available, the use of fire in combination with planting is advised. In unique situations where the loss of small intact natural areas—with an intact native seed bank—is unavoidable, the translocation of topsoil in combination with planting can be successful, even with a moderate budget.

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Supporting Information

The following information may be found in the online version of this article:

Table S1. The effect of restoration treatment, years after treatment, and their interaction on native species richness.
Table S2. The effect of restoration treatment, years after treatment, and their interaction on alien species richness.
Table S3. The total number of alien plant species present within each restoration treatment during the first and the second year.
Table S4. The effect of the different restoration treatments on growth form.
Table S5. The effect of restoration treatment, years after treatment, and their interaction on planted species survival.
Table S6. The effect of restoration treatment, years after treatment, and their interaction on sowed species survival.
Table S7. The estimated cost of the prescribed burn conducted for restoration treatments Burn, Burn-sow and Burn-sow-plant.
Table S8. The estimated cost of the prescribed burn conducted for restoration treatments Burn, Burn-sow and Burn-sow-plant.
Table S9. The estimated costs of the prescribed burn conducted for restoration treatments.
Table S10. The cost of each restoration treatment factor for the actual area size treated, as well as the cost scaled to 1 ha.
Table S11. The estimated costs of collecting, preserving, treating, transporting and sowing seed for restoration treatments Burn-sow, Burn-sow-plant, Remove-grass-sow-plant, and Soil-mulch-sow-plant.
Table S12. The estimated costs of collecting plant cuttings as well as propagating, transporting and planting out of the resultant plants for restoration treatments Burn-sow-plant, Remove-grass-sow-plant, Soil-mulch-sow-plant, and Soil–plant.
Table S13. The estimated costs of topsoil translocation for the restoration treatments Soil–plant and Soil-mulch-sow-plant.
Table S14. The estimated costs of mulch processing and transfer for the restoration treatment Soil-mulch-sow-plant.
Table S15. The estimated costs of the removal of alien grass layers with the top 10 mm of soil, as well as the removal of this material from the site, for the restoration treatment Remove-grass-sow-plant.

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