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RESEARCH ARTICLE

Stacking and Burning Invasive *Acacia saligna* Trees in South African Fynbos: Can Native Plant Species Passively Recover?

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ABSTRACT

The management of invasive taxa is crucial to mitigating their negative impacts and facilitating native biodiversity recovery. A common feature of clearing invasive trees in fire-prone systems is the 'stack burning' method. The intense and long duration of the heat experienced during burning, with areas at the centre of the stack reaching over 300°C and 175°C at the edge, often leaves persistent stack burn scars, limits native biodiversity recovery, facilitates secondary invasion, and alters soil biotic, chemical and physical conditions. Despite such negative impacts, stack burning remains a method for clearing invasive plants because it is an economical way to dispose plant biomass. Our study investigated the recovery of native species in 80 stack burn scars in lowland fynbos (Cape Flats Sand Fynbos) and mountain fynbos (Peninsula Sandstone Fynbos) of South Africa over 3 years after clearing. Our results indicate that the (1) native species recovery in stack burn scars was limited and (2) stack burn scars showed higher native species richness and cover in lowland than mountain fynbos. To encourage the recovery of native soil seed banks and biota in stack burn scars, active seeding and/or by covering them with topsoil is recommended.

1 | Introduction

Biological invasions exert significant negative ecological impacts on invaded ecosystems, disrupt ecosystem services, threaten human livelihoods and have led to significant global economic losses (Vilà et al. 2011; Simberloff et al. 2013; Roy et al. 2024). Management of invasive taxa is necessary to mitigate their impacts and facilitate native species recovery (Wittenberg and Cock 2005; Simberloff et al. 2013). Numerous biological, chemical and physical methods to clear invasive plants have been developed and applied globally (Messing and Wright 2006; Kettenring and Adams 2011; Weidlich et al. 2020).

A common feature of clearing efforts targeting invasive plants; particularly, in fire-prone systems where tree invasions are a major problem is the 'stack burning' method—that is, invasive plants are felled, their biomass is stacked and allowed to dry before burning (Korb, Johnson, and Covington 2004; Halpern, Antos, and Beckman 2014; Albrecht et al. 2022). Research shows that stack burning has negative ecosystem impacts (reviewed in Mott, Hofstetter, and Antoninka 2021) including persistent stack burn scars (Korb, Johnson, and Covington 2004; Fornwalt and Rhoades 2011), limited native biodiversity recovery in burnt areas (Fornwalt and Rhoades 2011; Creech, Katherine Kirkman, and Morris 2012; Halpern et al. 2012; Holmes, Esler, van

Wilgen, et al. 2020; but see Halpern, Antos, and Beckman 2014; Rhoades et al. 2015; Albrecht et al. 2022), facilitation of secondary invasion (Nsikani et al. 2020), and altered soil biotic, chemical and physical conditions (Haskins and Gehring 2004; Korb, Johnson, and Covington 2004; Creech, Katherine Kirkman, and Morris 2012; Holmes, Esler, van Wilgen, et al. 2020). Such impacts of stack burning are largely driven by the intense and long duration fires experienced during burning with areas at the centre of the stack reaching over 300°C in the centre (high severity fires) and 175°C at the edge (low severity fires) (Behenna, Vetter, and Fourie 2008; Owen et al. 2009). Despite the negative impacts, stack burning remains a preferred method for clearing invasive trees in fire prone systems because it is an economical method for the disposal of plant biomass (Owen et al. 2009; Mott, Hofstetter, and Antoninka 2021).

Negative effects of stack burning on soil conditions are well known as they have been researched for more than 50 years with emphasis on soil temperature, chemical and physical properties (Rhoades and Fornwalt 2015; Mott, Hofstetter, and Antoninka 2021). Over the last two decades, research on the effects of stack burning on native species recovery in various ecosystems has been on the rise (e.g., Korb, Johnson, and Covington 2004; DeSandoli, Turkington, and Fraser 2016; Albrecht et al. 2022). However, more research is needed on the impacts of stack burning and on post-burning native species recovery; particularly in ecosystems where this has not yet been studied, with emphasis on longer-term studies (Mott, Hofstetter, and Antoninka 2021). Management of plant biomass will remain critically important as the climate changes and fires become prevalent (Mott, Hofstetter, and Antoninka 2021; Nolan et al. 2021; Jones et al. 2022).

Acacia saligna (Labill.) H.L Wendl. (Fabaceae) has been purposefully introduced to many countries globally, primarily for site rehabilitation, fuelwood and charcoal production, and its invasions have been observed in Spain, South Africa, Portugal, Morocco, Kenya, Italy, Israel, Cyprus, Chile and Algeria (Richardson, Le Roux, and Marchante 2023). Approximately 600,000 ha are invaded by *Acacia saligna* globally, with 53,000 ha of this in South Africa, mostly in the fynbos biome (Griffin et al. 2011; Le Maitre et al. 2011; van Wilgen et al. 2011). Large-scale management efforts targeting *A. saligna* and other invasive taxa in South Africa have been on-going for almost three decades (van Wilgen, Wannenburgh, and Wilson 2022) and stack burning is widely used as a control method (particularly, in the fynbos) with cleared sites left to recover naturally (Blanchard and Holmes 2008; Holmes, Esler, Gaertner, et al. 2020). Other available control methods include: (1) Fell Only (trees cut and stumps treated with herbicide if species coppices); (2) Fell and Burn (without stacking); (3) Burn Standing (untreated trees burnt in a prescribed fire); (4) Fell and Remove (trees cut, stumps treated with herbicide if species coppices and larger wood removed from site); (5) Kill standing (trees ringbarked or frilled to kill roots and ultimately whole plant); (6) delay or avoid prescribed burning (Ngwenya et al. 2023, 2024) and (7) Aerial spraying (trees sprayed from above with herbicide by helicopter or boom-sprayer) (Parker-Allie et al. 2004; Holmes, Esler, van Wilgen, et al. 2020). It is worth noting that many factors determine whether mechanical clearing and the application of fire are successful in clearing *Acacia saligna* stands (Holmes, Esler, van Wilgen, et al. 2020).

Several factors influence restoration outcomes after clearing invasive plants and these include: (1) species of invader, (2) density of invasion, (3) duration of invasion, (4) ecosystem type and (5) method and efficacy of invasive species control (Holmes, Esler, van Wilgen, et al. 2020). Furthermore, whether passive or active restoration will lead to improved restoration outcomes depends on whether biotic and/or abiotic thresholds have been crossed (Holmes, Esler, van Wilgen, et al. 2020; Retief, Nsikani, and Geerts 2024). Ecosystems can recover passively, provided that key biotic and/or abiotic thresholds have not been crossed. However, if thresholds have been crossed, active restoration becomes necessary (Holmes, Esler, van Wilgen, et al. 2020).

Our study addressed the following: (1) does native species richness and cover after clearing *A. saligna* using the stack burning method differ between areas that had high and low severity fires (centre and edge of the stack burn scar respectively) and no fires (nearby unburnt areas), and does this change with years after clearing? (2) Does native species richness and cover after clearing *A. saligna* using the stack burning method differ in and between mountain and lowland fynbos, and does this change with years after clearing?

2 | Material and Methods

2.1 | Study Sites

Our study sites were in the Cape Flats Sand Fynbos (lowland) and Peninsula Sandstone Fynbos (mountain) (Rebelo et al. 2006). Native vegetation in the fynbos is typically evergreen shrublands which comprise ericoid, proteoid and restioid growth forms (Rebelo et al. 2006). The fynbos biome is regulated by natural fires which occur at intervals between 5 and 30 years and soils are largely nutrient-poor with Cape Flats Sand Fynbos soils comprising quaternary sand and Peninsula Sandstone Fynbos comprising colluvial sandy loam (Holmes 2002; Rebelo et al. 2006; Geerts 2021). Lowland fynbos is less resilient to invasion, and has a lower capacity for passive restoration, than mountain fynbos (Holmes, Esler, van Wilgen, et al. 2020). Blaauwberg Nature Reserve (33°46′5.16″ S; 18°27′10.08″ E) was the chosen study site for the Cape Flats Sand Fynbos. Glencairn (34°09′24.7″ S; 18°24′30.1″ E) was the chosen study site for the Peninsula Sandstone Fynbos (South African National Biodiversity Institute 2006). 40 ha of each study site was cleared using the stack burning method and was thus sampled in the study. During the study period (2014–2016), 266 mm in mean annual rainfall was received by Blaauwberg Nature Reserve and Glencairn received 775 mm (Nsikani et al. 2019). Before clearing in 2013, the study sites had seen *A. saligna* (75%–100% cover) invasions for more than 20 years.

2.2 | Study Design

The study design follows that in Nsikani et al. (2019). In September 2014, in each study site, we randomly chose 40 stack burn scars as far apart as possible ($n=80$). A transect was set-up in a south-east direction from the centre of each burn scar to the edge and outside of the burn scar. Areas sampled outside the burn scar did not experience burning, but were cleared of *A. saligna*, and were included to serve as a benchmark for recovery trajectories. Following this transect, 1 m^2 permanent plots were set-up at the centre, edge, and outside of each burn scar. The distance between the plots in each stack burn scar was equal. Differences in fire severity were not directly tested in this study since previous studies have already indicated that areas at the centre of the stack experience high severity fires and areas at the edge of the stack experience low severity fires (Behenna, Vetter, and Fourie 2008; Owen et al. 2009). All species present in each plot were recorded and their percentage cover estimated. Surveys were also conducted in September 2015 and 2016. There were follow-up clearing treatments targeting re-emerging *A. saligna* at 6-month intervals. We excluded secondary invaders (non-target alien species that proliferate following efforts to suppress dominant target invaders; Pearson et al. 2016) from this study as they were part of an earlier study (Nsikani et al. 2019).

2.3 | Statistical Analyses

Native species (Appendix A) cover in each plot was converted from percentages to proportions (e.g., 10% = 0.1; 50% = 0.5). Generalised linear mixed models (*glmmTMB* package; Magnusson et al. 2020) were used to investigate the influence of (1) *fire* (low and high severity fire, and no fire), (2) *fynbos type* (lowland and mountain), (3) *years after clearing* (one, two and three) and (4) the two-way interactions of the abovementioned factors on richness (log link function and Poisson error distribution) and cover (logit link function and Beta error distribution) of native species. 'Plot' was added to the models as a random effect. Dredging (*MuMIn* package; Barton 2018) was used to compare different models to determine the variables that influence richness and cover of native species. Information theoretic model procedures were used to compare models according to Akaike's information criteria (Burnham, Anderson, and Huyvaert 2011). To obtain relative parameter importance, estimates and confidence intervals, model averaging was performed on the most influential models (95% model confidence). Tukey's HSD test was used to highlight mean differences that were significant (*emmeans* package; Lenth 2018). R version 4.3.1 was used for all statistical analyses (R Core Team 2023).

3 | Results

We found 91 native plant species belonging to 65 genera and 33 families in the field throughout our study period (Appendix A). Families with the most species in our study were Asteraceae (21), Poaceae (nine), Cyperaceae (eight) and Scrophulariaceae (five). Genera with the most species in our study were *Helichrysum* (six), *Oxalis* (four), *Ficinia* (four) and *Ehrharta* (four) (Appendix A). *Senecio pterophorus* was present in all plots throughout the study period (Appendix A).

3.1 | Effect of Stack Burning on Native Species Richness and Cover

There was a significant interaction between the influence of *fire* and *years after clearing* on native species richness in the third year for low severity fires ($p = 0.032$; 95% confidence interval = −0.570 to −0.026). A similar relationship was observed, in the second ($p = 0.037$; 95% confidence interval = −0.544 to −0.016) and third ($p = 0.002$; 95% confidence interval = −0.686 to −0.158) year for no fire. In the first year, native species richness in areas that had high severity fires (1.9) was significantly lower than that in areas that had low severity fires (3.1) and no fire (3.7) (species richness was not significantly different in low severity fires and no fire) (Figure 1). However, this effect dissipated in the second year as all areas had similar native species richness, and this remained the case in the third year (Figure 1). Native species richness significantly increased in areas that had high and low severity fires and no fires in the second year (in relation to the first year) but remained the same in the third year (Figure 1).

There was a significant interaction between the influence of *fire* and *years after clearing* on native species proportional cover in the second ($p = 0.009$; 95% confidence interval = −0.961 to −0.138) and third ($p < 0.001$; 95% confidence interval = −1.197 to −0.395) year for no fire. In the first year, native species proportional cover in areas that had high severity fires (0.056) was similar to that in areas that had low severity fires (0.109), but both were significantly lower than that in areas that had no fire (0.226) (Figure 2). In the second and third year, while native species proportional cover in areas that had high severity fires remained significantly lower than that in areas that had no fire, native species proportional cover in areas that had low severity fires remained intermediate between that in the other two areas but not significantly different (Figure 2). Native species proportional cover significantly increased in areas that had high and low severity fires and no fires in the second year and afterwards in the third year (Figure 2).

3.2 | Effect of Fynbos Type on Native Species Richness and Cover

There was a significant interaction between the influence of *fynbos type* and *years after clearing* on native species richness in the second ($p = 0.002$; 95% confidence interval = −0.520 to −0.121) and third ($p < 0.001$; 95% confidence interval = −0.715

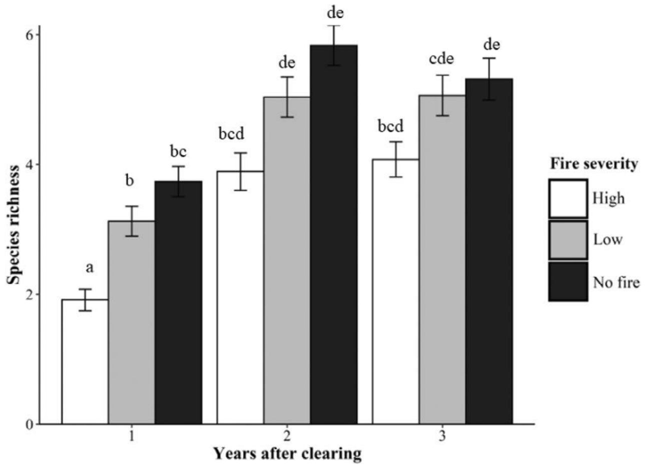

FIGURE 1 | Native species richness in areas that had no fires, low and high severity fires after clearing *Acacia saligna* in the fynbos. Values with the same letter are not significantly different. Error bars indicate ± SE.

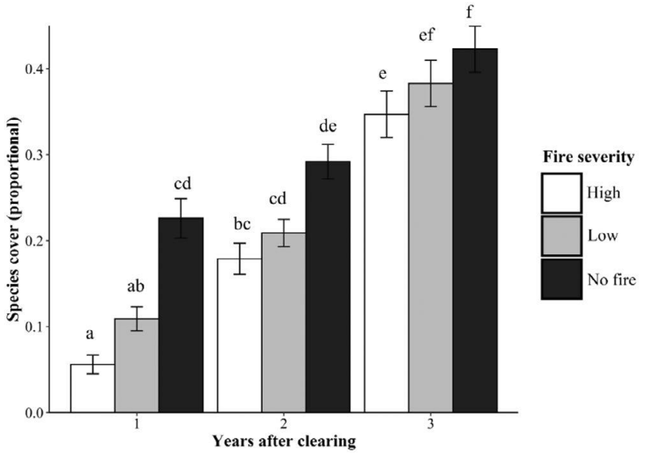

FIGURE 2 | Native species proportional cover in areas that had no fires, low and high severity fires after clearing *Acacia saligna* in the fynbos. Values with the same letter are not significantly different. Error bars indicate ± SE.

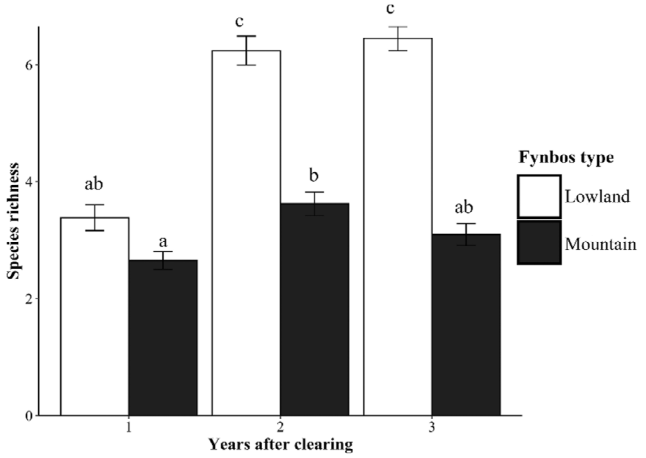

FIGURE 3 | Native species richness in mountain and lowland fynbos after clearing *Acacia saligna*. Values with the same letter are not significantly different. Error bars indicate ± SE.

to −0.310) year for mountain fynbos. In the first year, native species richness was similar between mountain and lowland fynbos (Figure 3). However, in the second and third year native species richness was significantly higher in lowland fynbos than in mountain fynbos (Figure 3). In both fynbos types, native species richness was significantly higher in the second year than in the first year and then remained the same in the third year (Figure 3).

There was a significant interaction between the influence of *fynbos type* and *years after clearing* on native species proportional cover in the second ($p < 0.001$; 95% confidence interval = −1.062 to −0.399) and third ($p = < 0.001$; 95% confidence interval = −1.492 to −0.849) year for mountain fynbos. Native species proportional cover was not significantly different between mountain and lowland fynbos in the first and second year (Figure 4). In the third year, native species proportional cover was significantly higher in lowland (0.467) than mountain (0.297) fynbos. In lowland fynbos, native species proportional cover significantly increased with each year after clearing, while in mountain fynbos native species proportional cover was

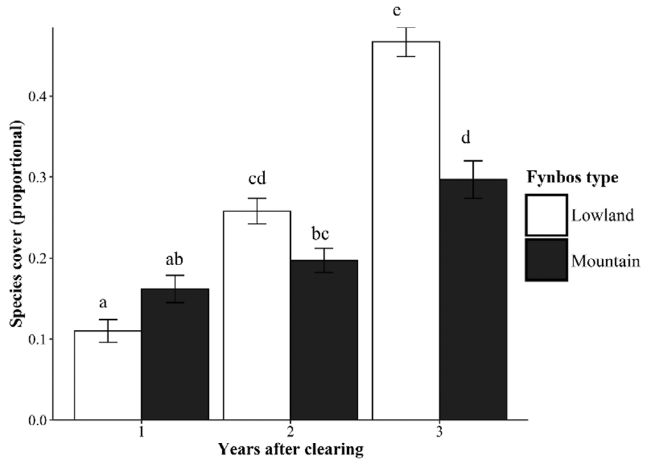

FIGURE 4 | Native species proportional cover in mountain and lowland fynbos after clearing *Acacia saligna*. Values with the same letter are not significantly different. Error bars indicate ± SE.

similar for the first 2 years and only increased significantly in the third year (Figure 4).

4 | Discussion

Our results suggest that 1 year after clearing, areas that had high severity fires have lower native species richness than those that had low severity fires or no fire. However, this effect dissipates in the second year as native species richness in areas that had high severity fires recover to resemble those that had low severity fires or no fires. There is no delay in the recovery of native species richness in areas that had low severity fires. Intense fires often lead to reduced native species richness across a range of ecosystems, particularly in the centre of the burn scar where heat is most intense (e.g., Fornwalt and Rhoades 2011; Creech, Katherine Kirkman, and Morris 2012; Mott, Hofstetter, and Antoninka 2021). High severity fires degrade native soil seedbanks by destroying or reducing the viability of seeds located in the upper soil horizons with often little or no viable seedbank remaining to regenerate (Korb, Johnson, and Covington 2004; Halpern et al. 2012; DeSandoli, Turkington, and Fraser 2016; Holmes, Esler, van Wilgen, et al. 2020). Furthermore, high severity fires often alter soil structure (Neary et al. 1999; Mott, Hofstetter, and Antoninka 2021) leading to negative effects on germination (Cilliers et al. 2004). Water-holding capacity and soil moisture significantly decrease in areas that had high severity fires leading to reduced germination because of reduced water availability (Neal, Wright, and Bollen 1965; Neary et al. 1999). Such water loss is driven by decreased soil permeability and organic layer loss due to the high heat (Neal, Wright, and Bollen 1965). With time, native species richness in areas that experienced high severity fires can increase to levels similar to unburnt areas as seen in our study due to species establishment from seeds that survived and/or colonised from nearby areas (Fornwalt and Rhoades 2011). Native species richness in areas that had low severity fires likely remained at similar levels to those in areas that had no fire in our study due to the lower heat that allowed for establishment of native species from surviving soil seedbanks (Fornwalt and Rhoades 2011). In fact, germination rates can be increased by the comparative low heat if the seeds are fire cued, and seeds in the fynbos are fire cued (Korb, Johnson, and Covington 2004; Behenna, Vetter, and Fourie 2008; Halpern, Antos, and Beckman 2014; Holmes, Esler, van Wilgen, et al. 2020).

Our results further indicate that it only takes 2 years for native species cover in areas that had low severity fires to be similar to that in areas that had no fires. This indicates that full recovery of stack burn scars remains limited up to 3 years after clearing. Similar results have been observed in studies of similar or longer durations (e.g., Fornwalt and Rhoades 2011; Creech, Katherine Kirkman, and Morris 2012; Halpern et al. 2012); however, the opposite has also been observed (Halpern, Antos, and Beckman 2014; Rhoades et al. 2015; Albrecht et al. 2022). Research on stack burning has shown that the recovery of vegetation is dependent on the ecosystem (Mott, Hofstetter, and Antoninka 2021). High severity fires during stack burning can alter the soil microbial composition leading to negative consequences on native plants (Neary et al. 1999; Haskins and Gehring 2004; Owen et al. 2009). For example, Korb, Johnson, and Covington (2004) highlight that stack burning almost eradicated arbuscular mycorrhizae which are crucial for the success of native plants in a ponderosa pine forest. Within one season, arbuscular mycorrhizae were showing signs of recovery, but only at the edge of the scar (Korb, Johnson, and Covington 2004). Arbuscular mycorrhizae hyphae likely persist in roots of plants that are in proximity to the burn or as mycorrhizal propagules deeper in the soil profile where there is less heat in areas that underwent low severity fires (Pattinson et al. 1999; Korb, Johnson, and Covington 2004). The success of native fynbos species has been shown to be limited by the absence of viable soil microbial communities (Nsikani et al. 2018). It is likely that in our study, soil microbial communities were altered in areas that had high severity fires, and this had a negative effect on native species establishment. In areas that suffered low severity fires, soil microbial communities could have recovered in the second year after clearing leading to native species cover being similar to that in unburnt areas. Changes in soil chemical composition such as elevated nitrogen availability that are brought about by stack burning (Neary et al. 1999; Korb, Johnson, and Covington 2004; Wolfson et al. 2005) are unlikely to have affected native fynbos species in our study as they have been shown to be relatively resistant to changes in soil chemical composition (Nsikani et al. 2018).

Our results highlight that the recovery trajectory of native species richness and cover differs between mountain and lowland fynbos. Holmes, Esler, van Wilgen, et al. (2020) highlight that lowland fynbos is less resilient to invasion and has a lower capacity for passive restoration. This is due to lower persistence of native seeds in the soil seed bank of lowland fynbos in comparison to mountain fynbos, and an even lower persistence of the longer-lifespan re-seeder species in lowland fynbos (Holmes, Esler, van Wilgen, et al. 2020). Holmes, Esler, van Wilgen, et al. (2020) propose that the lower persistence of native seeds in the soil seed bank of lowland fynbos is driven by higher fossorial mammal disturbance and granivore activity. Our results however indicate a lower capacity for passive restoration in mountain fynbos compared with lowland fynbos. Significant differences in native species richness between the two fynbos types sampled in our study were observed in the second and third year with lowland fynbos having a higher native species richness than mountain fynbos. Significant differences in native species cover between the two fynbos types included in our study were observed in the third year after clearing with lowland fynbos having higher native species cover than mountain fynbos. We acknowledge that we sampled only one site (covering 40 ha) in each fynbos type; therefore, caution is encouraged when interpreting our results due to the limited extent of our study.

Different recovery trajectories have been observed in different ecosystems after stack burning with some showing limited recovery (e.g., Covington, DeBano, and Huntsberger 1991; Owen et al. 2009; Creech, Katherine Kirkman, and Morris 2012) while others show quick recovery (e.g., Halpern, Antos, and Beckman 2014; Rhoades et al. 2015; Albrecht et al. 2022). It is likely that differences in resiliency between lowland and mountain fynbos, mostly due to differing community composition and physical environment accounted for the differences observed in native species recovery in our study (Selmants and Knight 2003; Holmes, Esler, van Wilgen, et al. 2020). Furthermore, the lowland fynbos sampled in our study could have had a higher soil seed bank seed persistence than mountain fynbos due to lower granivore activity and fossorial mammal disturbance (Holmes, Esler, van Wilgen, et al. 2020). It is also likely that the higher native species richness observed in lowland fynbos than mountain fynbos in our study was a result of species that had not established in the restoration site but occurred in surrounding areas, dispersing into the site (Fornwalt and Rhoades 2011). The higher native species richness in lowland fynbos in the second year could have led to the higher native species cover in the third year because of the additional species growing with time.

We acknowledge that stack burning is widely viewed as an economical method for disposing of plant biomass (Haskins and Gehring 2004; Korb, Johnson, and Covington 2004; Halpern, Antos, and Beckman 2014). However, wherever possible, we encourage the use of other methods instead of stack burning (Owen et al. 2009; Kraaij, Geerts, and Malan 2024). For example, invaders can be cleared, and the biomass removed without burning (Blanchard and Holmes 2008). Alternatively, the biomass can be removed and burnt on existing roads (Korb, Johnson, and Covington 2004). Where stack burning is unavoidable, there may be need to (1) apply seed to stack burn scars to improve native species recovery; or (2) burn stacks during the cooler season when soils are wet to minimise damage to seed banks and soils (although, germination cues for native seeds may not be met if the fires are not hot enough to trigger germination) (Korb, Johnson, and Covington 2004; DeSandoli, Turkington, and Fraser 2016; Holmes, Esler, van Wilgen, et al. 2020). Stack burn scars could also be covered with topsoil to encourage recovery of native soil seed banks and biota (Korb, Johnson, and Covington 2004; Owen et al. 2009; Albrecht et al. 2022). However, this is likely to be infeasible at large scales and to our knowledge has not been tested or trialled in South African fynbos.

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Conflicts of Interest

The authors declare no conflicts of interest.

Data Availability Statement

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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Appendix A

Native species found where there were high and low severity fires and no fires, up to 3 years after clearing invasive *A. saligna* in lowland and mountain fynbos. H = high severity fires; L = low severity fires; N = no fires. 1 = species present; – = species absent.

	First year after clearing						Second year after clearing						Third year after clearing					
	Lowland fynbos			Mountain fynbos			Lowland fynbos			Mountain fynbos			Lowland fynbos			Mountain fynbos		
Species	H	L	N	H	L	N	H	L	N	H	L	N	H	L	N	H	L	N
Albuca cooperi	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–	1	
Anthospermum prostratum	–	–	–	–	–	1	–	–	–	–	–	–	–	–	–	–	–	–
Arctotheca calendula	–	–	–	–	–		–	–	–	–	1	–	–	–	–	–	–	–
Aspalathus astroites	–	1		1	–	–	–	1	–	–	–	–	–	–	–	–	–	–
Asparagus ovatus	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–	–
Asparagus rubicundus	–	–	1	–	–	–	–	1	1	–	–	–	–	–	1	–	–	–
Bulbine alooides	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1
Carex aethiopica	–	–	–	1	–	–	–	–	–	1	–	–	–	–	–	–	–	–
Carpobrotus edulis	–	–	–	–	–	1	–	–	–	–	–	1	–	–	–	–	–	1
Centella triloba	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	1	1
Chironia baccifera	–	–	–	–	–	1	–	–	–	–	–	1	–	–	–	–	–	–
Cliffortia obcordata	–	–	–	–	–	–	–	–	1	1	–	–	–	–	–	–	–	–
Colchicum eucomoides	–	–	–	–	1	1	–	–	–	–	–	–	–	–	–	1	1	1
Colpoon compressum	–	–	–	–	–	1	–	–	–	–	–	1	–	–	–	–	–	–
Crassula campestris	1	1	1	–	–	–	1	1	1	–	1	1	1	1	1	1	–	–
Crassula glomerata	1	1	1	1	–	–	1	1	1	–	–	–	1	1	1	–	1	1
Cynodon dactylon	–	–	–	–	–	–	–	1	–	–	–	–	–	1	–	–	–	–
Dischisma ciliatum	–	–	–	–	1	1	–	–	–	1	1	1	–	–	–	1	1	1
Ehrharta calycina	1	1	1	1	1	1	1	1	1	–	–	–	1	1	1	–	–	–
Ehrharta longiflora	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	1
Ehrharta longifolia	–	–	–	–	1	1	–	–	–	–	–	–	–	–	–	–	–	–
Ehrharta villosa	–	1	1	1	1	1	–	1	1	1	1	1	1	1	1	1	1	1
Eriocephalus racemosus	–	–	–	–	–	–	–	–	–	–	–	–	1	1	1	–	–	–
Ficinia bulbosa	1	1	1	–	1	1	1	1	1	1	1	1	–	1	1	1	1	1
Ficinia deusta	–	1	1	–	1	1	–	–	–	–	–	1	–	–	–	–	–	1
Ficinia paradoxa	–	1	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Ficinia secunda	–	1	1	–	–	–	–	–	–	–	–	–	–	1	1	–	1	1
Galenia africana	–	–	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Gymnodiscus capillaris	1	1	1	–	–	–	1	1	1	–	–	–	1	1	1	–	–	–
Helichrysum cooperi	–	–	–	–	–	–	1	1	1	–	–	–	–	–	–	–	–	–
Helichrysum crispum	–	–	1	1	–	1	–	–	–	1	1	1	1	1	1	1	1	1
Helichrysum dasyanthum	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–
Helichrysum foetidum	–	–	–	–	–	–	–	–	–	–	–	–	1	1	–	–	–	–
Helichrysum moeserianum	1	1	1	1	1	–	1	1	1	1	1	1	1	1	1	1	1	1
Helichrysum patulum	–	–	–	1	1	1	–	–	–	–	–	–	–	1	–	–	–	1
Heliophila refracta	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–

Appendix A (Continued)

Species	First year after clearing						Second year after clearing						Third year after clearing					
	Lowland fynbos			Mountain fynbos			Lowland fynbos			Mountain fynbos			Lowland fynbos			Mountain fynbos		
	H	L	N	H	L	N	H	L	N	H	L	N	H	L	N	H	L	N
Hellmuthia membranacea	–	–	–	–	1	1	–	–	–	–	–	–	–	–	–	–	–	–
Hermannia hyssopifolia	–	–	–	–	1	1	–	–	–	–	–	–	–	–	–	–	–	–
Indigofera incana	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–	–
Isolepis cernua	–	–	–	1	1	1	–	–	–	–	–	–	–	–	–	–	–	–
Isolepis marginata	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Lampranthus falciformis	–	–	–	–	–	–	–	–	1	–	–	–	–	–	–	–	–	–
Leucadendron coniferum	–	–	–	–	1	1	–	–	–	–	1	1	–	–	–	–	–	1
Lyperia antirrhinoides	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–
Manulea tomentosa	–	–	–	–	–	–	–	–	–	–	–	–	1	1	1	–	–	–
Mesembryanthemum crystallinum	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	1	–
Metalasia densa	1	1	1	1	1	1	–	1	1	1	1	1	–	1	1	1	1	1
Metalasia muricata	–	–	–	1	1	1	–	1	1	1	1	1	–	–	–	1	1	1
Moraea angulata	1	1	–	1	1	1	–	1	1	–	–	–	–	–	–	–	–	–
Nemesia barbata	–	1	1	–	1	1	1	1	1	–	1	1	1	1	1	1	1	–
Olea capensis	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–	–	–
Osteospermumincanum	–	–	–	–	1	1	–	–	–	–	–	–	–	–	–	–	–	–
Osteospermummoniliferum	–	–	–	–	1	1	–	–	–	1	1	1	–	–	–	1	1	1
Otholobium bracteolatum	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–	–
Otholobium fruticans	–	–	–	–	1	1	–	–	–	–	–	–	–	–	–	–	–	1
Oxalis obtusa	–	–	1	–	–	–	–	1	1	–	–	–	–	1	1	–	–	1
Oxalis pes-caprae	–	1	1	–	–	–	1	1	1	–	1	1	1	1	1	–	–	1
Oxalis purpurea	–	1	1	–	–	–	–	–	–	–	–	1	–	1	1	–	1	1
Oxalis versicolor	–	–	–	–	–	–	–	–	1	1	1	–	–	–	–	1	1	–
Passerina corymbosa	–	1	1	–	1	1	–	1	1	1	1	1	1	1	1	1	1	1
Pelargonium betulinum	–	–	–	–	1	1	–	–	–	1	1	1	–	–	–	1	1	1
Pelargonium capitatum	–	1	1	1	1	1	–	–	–	1	1	1	–	–	–	1	1	1
Pelargonium myrrhifolium	–	–	–	–	–	–	–	1	1	1	1	1	1	1	1	–	–	1
Pentameris airoides	1	1	1	–	–	–	–	–	–	1	1	1	1	1	1	1	1	1
Pentameris barbata	1	–	–	–	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Petalacte coronata	1	1	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Phylica cephalantha	–	1	1	–	–	–	–	1	1	–	–	–	1	1	1	–	–	–
Phyllopodium capillare	1	1	1	–	1	–	–	–	–	1	–	1	1	1	1	1	1	–
Polygala myrtifolia	–	–	–	–	1	1	–	–	–	–	–	1	–	–	–	–	–	1
Protea scolymocephala	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–
Pseudoselago spuria	–	–	1	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–
Salvia africana-caerulea	–	–	1	–	–	–	–	–	1	–	–	–	–	–	1	–	–	–
Satyrium coriifolium	–	–	–	–	–	1	–	–	–	–	–	1	–	–	–	–	–	–

(Continues)

Appendix A (Continued)

	First year after clearing						Second year after clearing						Third year after clearing					
	Lowland fynbos			Mountain fynbos			Lowland fynbos			Mountain fynbos			Lowland fynbos			Mountain fynbos		
Species	**H**	**L**	**N**	**H**	**L**	**N**	**H**	**L**	**N**	**H**	**L**	**N**	**H**	**L**	**N**	**H**	**L**	**N**
Searsia lucida	–	–	1	–	1	–	–	–	–	–	–	–	–	–	–	1	1	–
Senecio burchellii	1	1	1	1	1	–	1	1	1	–	–	1	1	1	1	–	–	–
Senecio elegans	–	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Senecio pterophorus	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
Seriphium cinereum	–	–	–	–	–	1	–	–	–	–	1	–	1	–	–	–	–	–
Silene undulata	–	1	1	–	–	–	–	1	1	–	–	–	1	1	1	–	–	–
Solanum africanum	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	–	–
Stenotaphrum secundatum	–	–	–	–	–	–	–	–	–	–	–	–	–	–	–	1	1	–
Thesium funale	–	–	–	–	–	1	–	–	–	–	–	–	–	–	–	–	1	–
Trachyandra ciliata	–	–	–	–	–	1	–	–	–	–	–	1	–	–	–	1	1	1
Tribolium uniolae	–	–	–	1	1	1	–	–	–	1	1	1	–	1	1	1	1	1
Trichocephalusstipularis	–	–	–	–	–	–	–	–	1	–	–	–	–	–	1	–	–	–
Ursinia anthemoides	–	–	1	–	–	–	1	1	1	–	–	–	1	1	1	–	–	–
Wachendorfia paniculata	–	–	–	–	–	–	–	–	–	–	1	–	–	–	–	–	1	1
Wahlenbergia capensis	1	1	1	–	–	–	1	1	1	1	1	1	1	1	1	–	–	1
Wahlenbergia tenella	–	–	–	–	1	–	–	–	–	–	–	1	–	–	–	–	–	–
Watsonia marginata	–	–	–	1	–	–	–	–	–	–	–	1	–	–	–	–	–	1
Zaluzianskya divaricata	–	–	–	–	–	–	1	1	1	–	–	–	–	1	1	–	–	–