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RESEARCH ARTICLE

Effectiveness of integrated shrub management and restoration techniques for rehabilitating shrub-encroached rangelands of South Africa

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Abstract

Background: Shrub encroachment (SE) drives rangeland degradation, threatening biodiversity and forage productivity. The reversal of the catastrophic effects of SE depends on the efficacy of restoration efforts. We evaluated the potential of manual removal of the shrub *Pteronia incana* (Burm.f.) DC. with or without reseeding mixtures of grasses (*Panicum maximum* Jacq. and *Eragrostis curvula* (Schrad.) Nees) on rangeland restoration at Peddie, Eastern Cape Province, South Africa.

Methods: The treatments were (1) uprooting and reseeding (UPR), (2) uprooting only (UPO), (3) cutting and reseeding (CR), (4) cutting only (CO) and (5) untreated control (CTL), each replicated three times in North and South aspects.

Results: Graminoid cover was higher in all treatments than in CTL, with CR showing relatively higher cover in North and South slopes. Restoration treatments increased plant density, richness and diversity compared to CTL. Slope aspect, treatment and seeded species interacted significantly on biomass production. CO in the North aspect attained two-fold higher biomass production of nonreseeded grasses than other treatments. Responses of reseeded species varied by aspect, with *P. maximum* attaining two-fold higher BP in CR and UPR in the North than the South slope, whereas *E. curvula* had higher biomass production in CR in the South than the North slope.

Conclusions: Shrub clearing facilitated restoration of diversity and plant densities regardless of the restoration method. However, biomass production and vegetation cover depended on the interaction of the shrub clearing method, reseeding and slope aspect. Thus, the decision on restoration of herbaceous vegetation should be informed by the vegetation attribute of restoration priority.

KEYWORDS

Eragrostis curvula, *Panicum maximum*, reseeding, shrub clearing, shrub encroachment

INTRODUCTION

Shrub encroachment (SE) refers to an increase in shrub abundance, cover and biomass at the expense of herbaceous vegetation (Archer et al., 2017). It is regarded as one of the forms of land degradation (Manjoro et al., 2012), causing decline in biodiversity, and herbaceous productivity, leading to soil erosion and reduced grazing potential of rangelands (Sholto-Douglas et al., 2017). This phenomenon has been reported in tropical, temperate and, to a certain degree in Arctic climatic zones across the globe for at least over 150 years in other regions (Graw et al., 2016; Grigoriev et al., 2022; Huang et al., 2018; O'Connor et al., 2014; Shen et al., 2022). Globally, about 500 million hectares of land is encroached by woody plants (Ding & Eldridge, 2024), with consequential decline in ecosystem services, including hydrology and energy balance. However, the extent of

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encroachment varies regionally, with semi-arid areas of southern Africa reported to be severely affected (Mndela, Madakadze, Tjelele, et al., 2022). For instance, an area of 10–20 million ha in South Africa, 26–30 million ha in Namibia and 3.7 million ha in Botswana are severely encroached by woody plants (Kellner et al., 2021; Kgosikoma, 2012; Shikangalah & Mapani, 2020; Stafford et al., 2017).

The effect of bush encroachment is reported to vary with the stature of the encroaching species, with dwarf shrubs causing more severe effects than tall encroachers due to the projection of their canopies close to the soil surface (Blaser et al., 2013). These shrubs intercept light more efficiently, thereby limiting the photosynthetic capacity and productivity of understorey light-dependent C_4 grasses (Pons & Poorter, 2014). This leads to the extinction of desirable herbaceous species (Charles-Dominique et al., 2018; Klopper, 2015). The prominent shrub encroachers in South Africa largely comprise dwarf shrubs of the Asteraceae family (Odindi, 2009). In the Thicket biome of the Eastern Cape Province, a shrub called *Pteronia incana* (Burm.f.) DC. is encroaching widely in communal rangelands (Kakembo et al., 2012; Mlamla et al., 2022). This biome is known for its high biodiversity status and provides other ecosystem services, including grazing, largely for communal livestock and game ranching (Hoare et al., 2006).

Vigorous encroachment of *P. incana* jeopardises these ecosystem services, more so given that this shrub is unpalatable to herbivores. This shrub usually grows in a steep southerly slope orientation, predominantly in drier upper slopes (Kakembo et al., 2007). It promotes the crusting of the topsoil and erosion, thus reducing the water infiltration rate. According to Tongway and Hindley (1995), this lowers soil nutrient cycling and reduces the stability of native herbaceous vegetation. Furthermore, *P. incana* has the ability to retain higher soil moisture content after rain than grasses, thus gaining an advantage of persisting longer than grasses during the drought periods (Odindi & Kakembo, 2011).

Apart from its competitive effect for resources, *P. incana* is reported to have phytotoxic compounds such as phenols, flavonoids, monoterpenes and so forth (Hulley et al., 2010), which are allelopathic to understorey vegetation (Lesoli et al., 2013). This shrub not only affects the above-ground herbaceous vegetation composition and cover but also has adverse effects on the soil seed bank composition and size, with serious ramifications for herbaceous recovery (Haussmann et al., 2019). Clearing of *P. incana*, therefore, calls for restoration measures that will ensure the successful recovery of herbaceous vegetation in shrub-encroached rangelands. However, according to Briske et al. (2005), reversal of shrub-encroached sites to their pre-encroachment state may not be feasible if the threshold of encroachment has been surpassed, in which case proactive woody plant management can be the only solution. However, it remains unclear if passive restoration of herbaceous vegetation via successional vegetation recovery following clearing of *P. incana* is effective or if active restoration is necessary in the Thicket biome. Furthermore, an effective shrub clearing technique for the restoration of *P. incana*-encroached rangelands has yet to be developed. Previous attempts have been made to restore *P. incana*-encroached rangelands via shrub clearing and or use of fire, with recovery depending on the passive regeneration of vegetation (Ntutha, 2016; Palmer, 2010). However, more often, the expected outcomes have never been realised due to a tendency of secondary invasion by undesirable plants (Palmer, 2010) and or *P. incana* re-encroachment from the seed bank (Haussmann et al., 2019).

Furthermore, passive restoration takes too long, approximately 10–20 years, for rangeland to revert to a pre-encroachment state (Chazdon et al., 2021; Zahawi et al., 2014). Thus, alternative restoration techniques incorporating shrub clearing and herbaceous plant reinforcement need to be explored. This includes reseeding highly adapted and productive herbaceous plants following shrub eradication. Moreover, cutting of woody plants promotes plant coppicing, whereas uprooting promotes the recruitment of annual plants, largely pioneer forbs and grasses (Castillo-Garcia et al., 2024; Suding et al., 2013). Thus, active restoration through reseeding perennial species can be a strategy to fast-track the vegetation recovery process (Castillo-Garcia et al., 2024).

Studies by Tebeje et al. (2014), Brancalion et al. (2016), Abdellatif et al. (2016) and Slim et al. (2021) showed that reseeding improves the species diversity, richness and productivity of rangelands within a short period of time. However, to the best of our knowledge, very few, if any, studies have assessed the efficacy of integrated shrub management and active restoration in rehabilitating herbaceous vegetation in *P. incana*-encroached rangelands. This information is needed not only for the restoration of diversity and productivity of rangelands but also for policy development. Thus, to bridge these knowledge gaps, we designed this study to answer the following questions: (1) Are the restoration treatments effective in restoring the herbaceous vegetation in *P. incana*-encroached rangelands? (2) Does the integration of shrub clearing and grass reseeding have more pronounced effects on the recovery of herbaceous plants than shrub clearing only? (3) Does the slope aspect influence the effectiveness of the restoration treatments on herbaceous plant recovery?

MATERIALS AND METHODS

Study site

The study was conducted in *P. incana*-encroached communal rangelands of Qeto Village in Ngqushwa district (formerly known as Peddie) of the Eastern Cape in South Africa (Figure 1). The village is located at an altitude of 224 m a.s.l. with coordinates of 33°19′ S, 27°06′ E along the eastern coast of South Africa. The area receives rainfall of 412 mm $year^{-1}$), and the rainfall is bimodal peaking around March–April 2023 and October–November 2023, with most in the presummer peak. The average temperature in summer (December–April) and winter (June–August) ranges from 9°C to 22°C and from 8°C to 17°C, respectively (Haussmann et al., 2019; Sifanele & Shackleton, 2025). The geology of the area

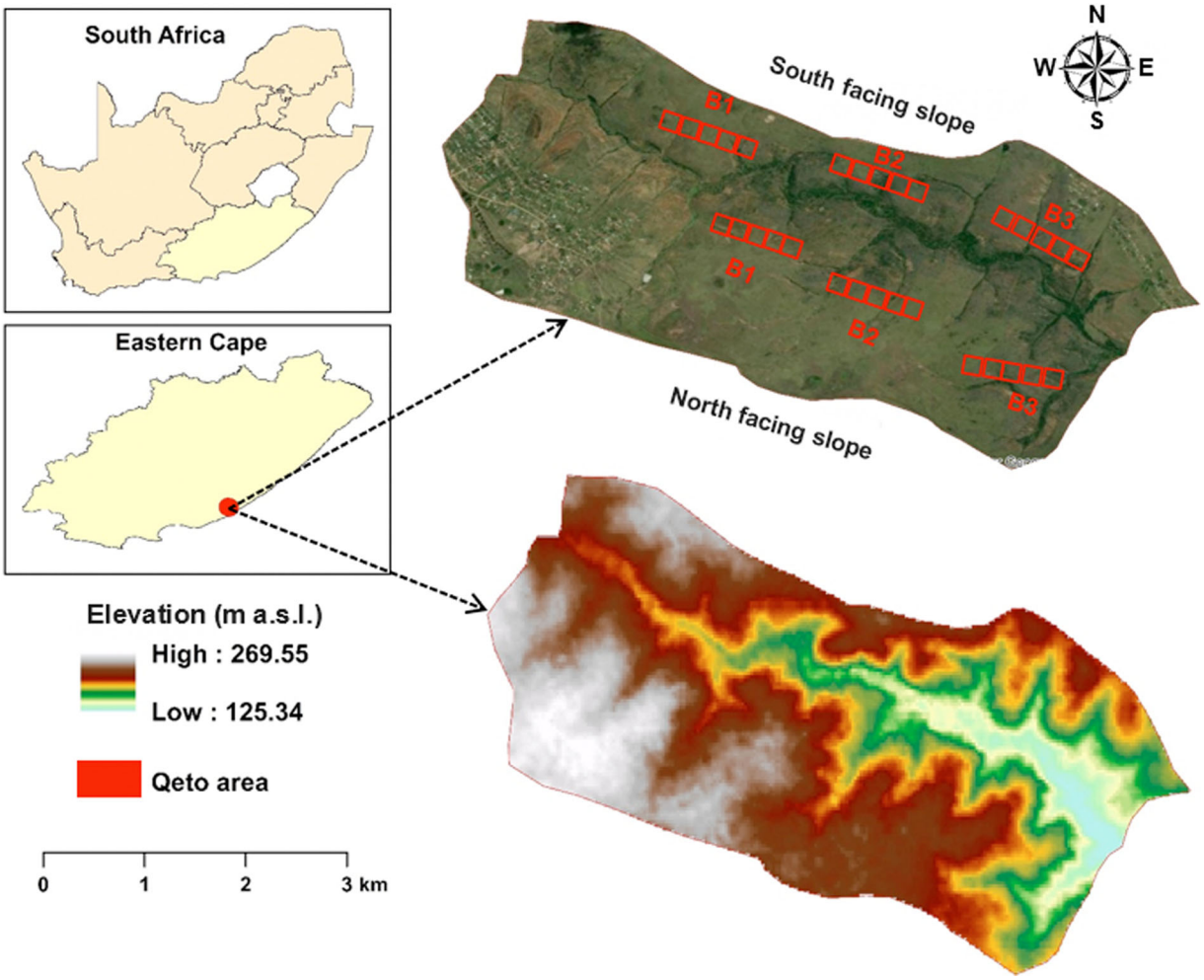


FIGURE 1 The study site (Qeto Village in Peddie) in the Eastern Cape Province of South Africa. In the map, B1, B2 and B3 denote Blocks 1, 2 and 3, respectively.

is primarily shales and red mudstones of the Ecca Group and Karoo Super Group (Manjoro et al., 2012), forming soils that are shallow and rocky with high swelling hydrous mica clays (Haussmann et al., 2019). Furthermore, the soils are also low in organic matter and prone to surface crusting and soil erosion (Kakembo, 2009).

Lower Qeto is located within the Thicket biome (Mucina & Rutherford, 2006). The vegetation is described as Eastern Thorn Bush-veld (Low & Rebelo, 1996). Woody vegetation comprises *Acacia karoo* Hayne, and thicket species (e.g., *Scutia myrtina* (Burm.f.) Kurz, *Searsia* spp.) that form bush clumps. The grass layer is dominated by *Themeda triandra* Forssk, *Sporobolus africanus* (Poir.) Robyns & Tournay and *Digitaria eriantha* Steud. (Ntutha, 2016). The rangeland is grazed rotationally by communal livestock.

Experimental design

This experiment comprised four restoration techniques and a control (CTL), namely, (1) uprooting of the shrub and grass reseeding (uprooting and reseeding [UPR]); (2) uprooting only, without reseeding (UPO); (3) cutting of the shrub and reseeding (CR); (4) cutting only without reseeding (CO); and (5) no shrub removal and reseeding (CTL). These treatments were applied in heavily encroached sites with ≥85% shrub cover in North- and South-facing slopes (Figure 2). In each slope aspect, a 1-ha area in fenced grazing camps was demarcated, in which each restoration method was applied in three replicate plots per slope aspect, totalling 30 plots (5 treatments × 2 slope aspects × 3 replicates). The replicate plots were 10 m × 10 m in size, separated by 10 m, with treatments arranged as a completely randomised block design. This experiment comprised three blocks, with each block assigned one replicate plot per treatment.

Shrub clearing and reseeding methods

The clearing methods used to remove *P. incana* on the experimental plots were uprooting and cutting. Uprooting was performed using a dicker, which uproots the plant with its roots, whereas cutting was conducted by slashing the shrub at the base approximately 6 cm above the soil surface using slashers. Following uprooting, the soil was levelled with a rake to ensure uniformity of the seedbed. Thereafter, a mixture of forage grasses (*Panicum maximum* Jacq, variety Gatton, and *Eragrostis curvula* [Schrad.] Nees, variety Ermelo) was planted



FIGURE 2 *Pteronia incana* (Burm.f.) DC.-invaded sites (a) before, (b, c) immediately after clearing and in (d) cutting only and (e) grass reseeded plots.

through broadcasting by hand, followed by rolling to cover grass seeds. The nonreseeded *P. maximum* and *E. curvula* were easily distinguishable from the reseeded ones based on morphological traits, for example, maximum height at maturity and leaf size. The grass species used in this study were selected, since they are native to the study area and are highly productive with good grazing value, grow well in slightly acidic soils and are drought-resistant (Van Oudtshoorn & Van Wyk, 1999). In CR, grass seeds were broadcast, followed by minimal soil disturbance. This was done to break the soil crust and increase seed–soil contact for better germination (Van Staden et al., 2018). The experiment was conducted in February 2023, and the data for herbaceous species composition, biomass and plant cover were collected 12 months later.

Data collection

Five 0.25 m^2 quadrats were sampled per plot in the four corners and centre of each experimental plot, totalling 150 quadrats (30 plots × 5 quadrants) for the experiment (Figure 2a). Plants were identified to species level and recorded in each quadrat, and the maximum and minimum basal diameters were measured using a standard ruler (Figure 2b). Herbaceous cover, including graminoids, nongraminoids and bare soil covers, was estimated

visually in each quadrat. Afterward, species frequencies and plant densities were computed. The herbaceous plants were classified into grasses, sedges, forbs, succulents and karoo shrubs. The grasses were also classified according to their ecological status (Decreasers, Increasers I and II), grazing value (high, moderate and low) and lifeform (annuals, biennials and perennials). Decreasers are defined as desirable species that dominate when the rangeland is properly managed or in excellent condition; Increaser I denotes less desirable species that are dominant when the rangeland is underutilised; and Increaser II denotes less desirable species dominant when the rangeland is overgrazed or overutilised (Mndela, 2013).

The collected data were used to estimate plant species density, richness and diversity. Species richness is the number of different types of species, whereas evenness is the proportion of individual species in the community (Tuomisto, 2012). Species diversity was determined using the Shannon–Wiener index or the Shannon diversity index (Nolan & Callahan, 2006) using the following equation:

$$H' = -\sum_{i=1}^{s}[(p_i) \times \ln(p_i)], \quad (1)$$

where H' = Shannon diversity index, p_i = proportion of individuals of ith species in the whole community, ln = natural logarithm, i = each species and S = number of species in the community.

The basal cover (BC) was computed from maximum and minimum canopy basal diameters of individual plants using the formula for the area of an ellipse (Jacobs & Schloeder, 2002).

$$\mathrm{BC} = \pi r_1 \times r_2, \quad (2)$$

where r_1 is the radius of the maximum diameter, and r_2 is the radius of the minimum diameter.

The area covered by individual plants was then summed up and converted into a percentage based on the area of a quadrat.

Biomass production

Herbaceous plants were harvested destructively at a stubble height of 5 cm in each 0.25 m^2 quadrat using a sheep shearing handpiece. During this process, the grasses were harvested by species and placed in different paper bags for each quadrat, whereas the nongrassy plants, including forbs and succulents, were harvested together and placed in a single paper bag. Then, plant samples were oven-dried at 60°C for 48 h in the laboratory. After oven-drying, the samples were weighed to determine dry matter (kg DM ha^{-1}).

Statistical analysis

Before data analysis, Shapiro–Wilk and Levene's tests were used to perform data normality and homoscedasticity tests, and variables that failed the test were transformed. Herbaceous plant biomass production was transformed using $\log(x + 1)$, and the data were back-transformed to their original form after the analysis. Analysis of variance (ANOVA) was conducted using general linear mixed models to determine the interactive and independent effects of slope aspect and restoration method on herbaceous vegetation covers, diversity, richness, density and biomass using Jamovi statistical software version 2.4.14.0. The block was included as a fixed factor, whereas the plot was included as a random factor in the model to serve as the error correction term. The statistical model used was as follows:

$$Y_{ijkl} = \mu + \beta_i + A_j + T_k + AT_{jk} + \varepsilon_{ijkl}, \quad (3)$$

where Y_{ijkl} = response variable, μ = overall mean, β_i = effect of the ith block ($n = 3$), A_j = effect of the jth slope aspect ($n = 2$), T_k = effect of the kth treatment ($n = 5$), AT_{jk} = interactive effect of the jth slope aspect and kth treatment ($n = 10$) and ε_{ijkl} = the random error ($n = 30$).

When the interaction between treatment and slope aspect was significant for the response variable at $\alpha = 5\%$, we reported only the interaction effect. Then, a post hoc analysis using the Tukey test was performed to conduct mean separation. A correlation matrix was compiled to determine the relationship (r) between herbaceous plant attributes at a 95% significance level.

RESULTS

Herbaceous species composition

In total, 19 herbaceous species were recorded; UPO in the North-facing slope had more herbaceous plant species ($n = 12$) compared to other treatments, with grasses being dominant (68.19%), followed by karoo shrubs (23.70%) and forbs (7.41%). *Aristida congesta* Roem. & Schult. (18.02%), *D. eriantha* (11.11%) and nonreseeded *P. maximum* (10.37%) were the most dominant grass species in UPO in the North-facing slope. Furthermore, *D. eriantha* was present across all treatments on the North aspect, whereas on the South aspect, it was only found in CR and UPR. However, in the South aspect, UPR showed more plant species ($n = 11$) than CTL ($n = 6$). Of these species, grasses and forbs dominated by 47.23% and 42.76%, respectively. Reseeded *E. curvula* was the most dominant grass species (17.28%) in the South-facing slope. *Karroochloa curva* (Nees) Conert & Türpe was only found in the South aspect across all restoration treatments. However, nonreseeded *P. maximum* and *Helictotrichon turgidulum* (Stapf) Schweick. were only found in the North aspect, and these grasses were more abundant in the CTL. *A. congesta*, nonreseeded *E. curvula* and *Eragrostis capensis* (Thunb.) Trin. increased with shrub clearing. The forbs were more abundant in CTL and reduced in shrub clearing treatments (Table 1).

Plant density, richness and diversity

The interaction between slope aspect and treatment was not significant ($p = 0.865$), but the treatment

TABLE 1 Relative abundance of herbaceous plant species under various restoration treatments for North (N) and South (S) slope aspects.

Plant species	Lifeform	GV	ES	Aspect	CTL	CO	CR	UPO	UPR
A. congesta	Perennial	Low	Increaser II	N	n.p.	11.08	14.97	18.02	n.p.
				S	n.p.	n.p.	n.p.	7.44	0.89
K. curva	Annual	Low	Increaser I	N	n.p.	n.p.	n.p.	n.p.	n.p.
				S	5.88	6.00	4.46	7.44	4.15
D. eriantha	Perennial	High	Decreaser	N	11.76	11.08	6.74	11.11	9.06
				S	n.p.	n.p.	7.81	n.p.	5.34
E. capensis	Perennial	Moderate	Increaser II	N	n.p.	5.17	2.99	4.44	n.p.
				S	n.p.	6.00	n.p.	1.24	n.p.
E. chloromelas	Perennial	Moderate	Increaser II	N	n.p.	n.p.	n.p.	4.44	7.05
				S	n.p.	n.p.	n.p.	7.44	n.p.
E. curvula	Perennial	High	Increaser II	N	n.p.	n.p.	21.96	n.p.	21.91
				S	n.p.	n.p.	22.26	n.p.	17.28
E. curvula (nonreseeded)	Perennial	High	Increaser II	N	n.p.	7.39	2.25	4.44	4.03
				S	n.p.	6.00	n.p.	5.58	1.78
E. plana	Perennial	Low	Increaser II	N	11.76	7.02	n.p.	7.90	n.p.
				S	n.p.	4.50	n.p.	9.92	3.56
H. contortus	Perennial	Moderate	Increaser II	N	n.p.	1.48	5.99	1.48	n.p.
				S	5.88	n.p.	n.p.	1.24	n.p.
H. turgidulum	Perennial	Moderate	Decreaser	N	17.65	n.p.	n.p.	n.p.	n.p.
				S	n.p.	n.p.	n.p.	n.p.	n.p.
P. maximum	Perennial	High	Decreaser	N	n.p.	n.p.	12.48	n.p.	12.29
				S	n.p.	n.p.	8.37	n.p.	4.89
P. maximum (nonreseeded)	Perennial	High	Decreaser	N	23.53	2.96	n.p.	10.37	n.p.
				S	n.p.	n.p.	n.p.	n.p.	n.p.
P. stapfianum	Perennial	High	Decreaser	N	n.p.	n.p.	n.p.	n.p.	n.p.
				S	n.p.	n.p.	n.p.	n.p.	6.23
S. fimbriatus	Perennial	High	Decreaser	N	n.p.	n.p.	4.49	6.67	n.p.
				S	n.p.	n.p.	n.p.	n.p.	n.p.
T. triandra	Perennial	High	Decreaser	N	11.76	3.94	n.p.	n.p.	n.p.
				S	5.88	3.00	2.79	2.48	n.p.
Forbs	–	Low	Increaser II	N	n.p.	26.60	7.49	7.41	26.19
				S	47.06	33.02	18.60	33.06	42.76
Karoo shrubs	–	Low	Increaser II	N	11.76	23.28	20.66	23.70	19.47
				S	n.p.	37.72	33.48	15.5	10.01
Sedges	–	Low	Increaser II	N	n.p.	n.p.	n.p.	n.p.	n.p.
				S	23.53	2.25	1.12	8.68	3.11
Succulents	–	Low	Increaser II	N	11.76	n.p.	n.p.	n.p.	n.p.
				S	11.76	1.50	1.12	n.p.	n.p.

Note: Full Latin names with taxonomic authority: *A. congesta, Aristida congesta* Roem. & Schult.; *K. curva, Karroochloa curva* (Nees) Conert & Türpe; *D. eriantha, Digitaria eriantha* Steud.; *E. capensis, Eragrostis capensis* (Thunb.) Trin.; *E. chloromelas, Eragrostis chloromelas* Steud.; *E. curvula, Eragrostis curvula* (Schrad.) Nees; *E. plana, Eragrostis plana* Nees; *H. contortus, Heteropogon contortus* (L.) P. Beauv. ex Roem. & Schult.; *H. turgidulum, Helictotrichon turgidulum* (Stapf) Schweick.; *P. maximum, Panicum maximum* (Jacq.); *P. stapfianum*, *Panicum stapfianum* Fourc.; *S. fimbriatus*, *Sporobolus fimbriatus* (Nees ex Trin.) Nees; *T. triandra*, *Themeda triandra* Forssk.

Abbreviations: ES, ecological status; GV, grazing value; n.p., not present. Aspect: N, North; S, South. Treatments: CO, cutting only; CR, cutting and reseeding; CTL, control; UPO, uprooting only; UPR, uprooting and reseeding.

TABLE 2 Analysis of variance (ANOVA) table showing *F* and *p* values of aspect, treatment, species or functional group and their interactions on herbaceous vegetation.

Response variable	Source of variation	df	*F*	*p*
Graminoid cover (%)	Block	2	3.21	0.077
	Slope aspect	1	21.67	**<0.001**
	Treatment	4	42.77	**<0.001**
	Aspect × Treatment	4	3.76	**0.007**
Nongraminoid cover (%)	Block	2	0.66	0.418
	Slope aspect	1	4.60e−5	0.995
	Treatment	4	79.55	**<0.001**
	Aspect × Treatment	4	1.42	0.235
Bare soil (%)	Block	2	0.04	0.834
	Slope aspect	1	14.71	**<0.001**
	Treatment	4	9.97	**<0.001**
	Aspect × Treatment	4	0.87	0.487
Litter cover (%)	Block	2	8.74	**0.004**
	Slope aspect	1	0.05	0.818
	Treatment	4	2.10	0.088
	Aspect × Treatment	4	0.78	0.541
Basal cover (%)	Block	2	0.66	0.798
	Slope aspect	1	48.73	**<0.001**
	Treatment	4	14.96	**<0.001**
	Aspect × Treatment	4	4.53	**0.002**
Plant density (plants m^{-2})	Block	2	0.025	0.876
	Slope aspect	1	2.902	0.119
	Treatment	4	20.96	**<0.001**
	Aspect × Treatment	4	0.310	0.865
Species richness	Block	2	0.00	1.00
	Slope aspect	1	13.44	**0.004**
	Treatment	4	16.78	**<0.001**
	Aspect × Treatment	4	2.19	0.143
Species diversity	Block	2	0.02	0.887
	Slope aspect	1	2.19	0.169
	Treatment	4	17.48	**<0.001**
	Aspect × Treatment	4	2.04	0.164
Biomass	Block	2	0.15	0.697
	Aspect	1	1.77	0.185
	Treatment	4	32.55	**<0.001**
	Species name	3	57.806	**<0.001**
	Aspect × Treatment	4	0.353	0.842
	Aspect × Species/PFG	3	5.427	**0.001**
	Treatment × Species/PFG	12	12.372	**<0.001**
	Aspect × Treatment × Species/PFG	12	1.934	**0.029**

Note: The bold *p* value denotes a significant difference at $p \leq 0.05$.

Abbreviation: PFG, plant functional group.

independently had a significant effect on plant density ($p < 0.001$) (Table 2). Plant density in UPR was comparable ($p > 0.05$) to CR and UPO but differed significantly from CO ($p = 0.029$) and CTL ($p < 0.001$). UPR had the highest plant density (217 ± 14.8 plants m^{-2}), whereas CTL had the lowest density (26 ± 14.8 plants m^{-2}) (Figure 3a).

For species richness, no significant interaction between slope aspect and treatment was observed ($p = 0.143$), but slope aspect ($p = 0.004$) and treatment ($p < 0.001$) independently affected species richness (Table 2). The restoration treatments had similar ($p > 0.05$) species richness, with CTL having five to seven species less than in other restoration treatments (Figure 3b).

The treatments significantly affected ($p < 0.001$) species diversity (Table 2). The species diversity responses resembled those of species richness, ranging from 1.19 ± 0.09 to 2.15 ± 0.09, with all restoration methods having significantly higher diversity compared to the CTL (1.19 ± 0.09). UPO, in particular, increased species diversity to values two-fold higher than the CTL (Figure 3c).

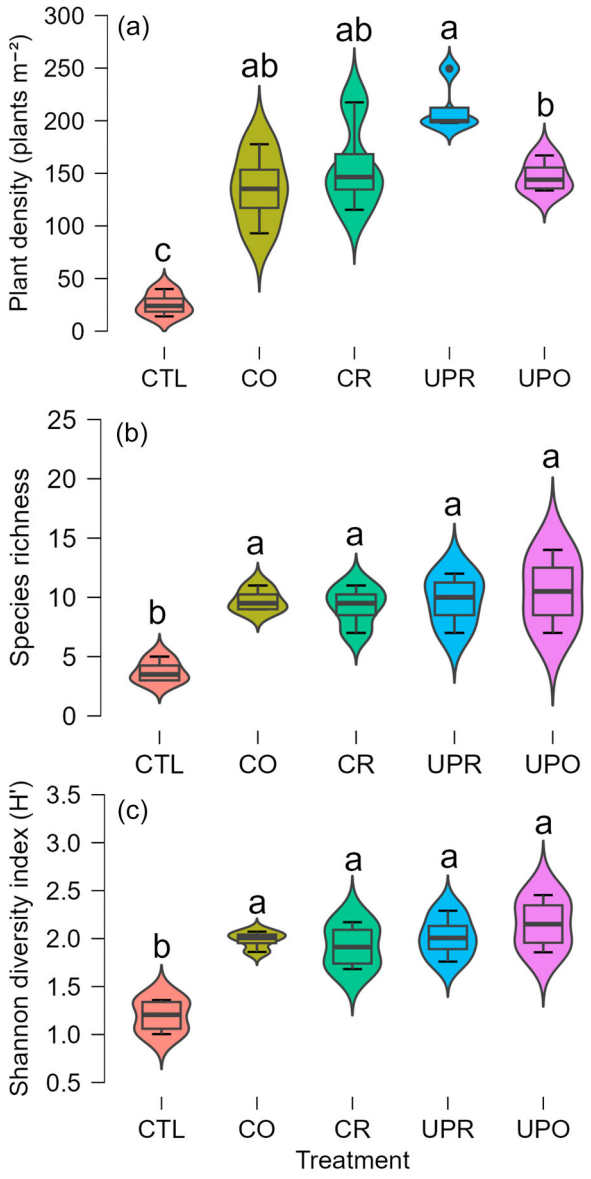


FIGURE 3 (a) Herbaceous plant density, (b) richness and (c) diversity across different restoration treatments. The line inside the box shows the median. The lower (25%) and upper (75%) quartiles are represented by the lower and upper limits of the box, whereas the minimum and maximum values are represented by the lower and upper whiskers, respectively. CO, cutting only; CR, cutting and reseeding; CTL, control treatment; UPO, uprooting only; UPR, uprooting and reseeding. Different superscripts (a, b, c) indicate significant differences between treatments.

Vegetative and nonvegetative covers

The interaction between slope aspect and treatment ($p = 0.007$), slope aspect ($p < 0.001$) and treatment ($p < 0.001$) was significant for graminoid cover (Table 2). All treatments had significantly ($p < 0.001$) increased graminoid cover compared to the CTL in the North-facing slope. In the South-facing slope, treatments had a similar effect to that on the North aspect, except for UPO, which did not improve graminoid cover significantly ($p = 0.058$) compared to the CTL. UPO in the North aspect had a two-fold higher graminoid cover compared to UPO in the South aspect ($p < 0.001$, Figure 4a).

The two-way interaction between slope aspect and treatment on the nongraminoid cover was not significant ($p = 0.235$), nor was the effect of slope alone ($p = 0.995$). However, restoration treatments ($p < 0.001$) significantly affected nongraminoid cover (Table 2), with CTL in the North and South aspects achieving significantly higher cover ($p < 0.05$) four to eight times greater than that in other restoration treatments (Figure 4b). For bare soil cover, the interaction between the slope aspect and treatment had no significant effect ($p = 0.487$), but the slope aspect and treatment independently had a significant effect ($p < 0.001$, Table 2). UPO and CO in the South-facing slope had two- to three-fold higher incidence of bare patches than UPR, CR and CTL in the North slope (Figure 4c).

The interactive effects between slope aspect and treatment were significant for BC ($p = 0.002$, Table 2). Generally, the BC was relatively low, ranging from $0.35\% \pm 0.97\%$ to $3.48\% \pm 0.97\%$ in the South aspect and from $0.38\% \pm 0.97\%$ to $10.95\% \pm 0.97\%$ in the North aspect. There were no noticeable differences between treatments in the South slope ($p > 0.05$). However, all restoration treatments had 12- to 31-fold higher BC relative to the CTL in the North slope. Particularly, CO treatment attained a two-fold higher BC compared to the UPR treatment ($p = 0.005$, Figure 4d).

Biomass production

A significant three-way interaction of slope aspect, treatment and species ($p = 0.029$) was observed for herbaceous plant biomass (Table 2). The biomass of nonreseeded grasses in the CO treatment (1820 ± 122 kg DM ha^{-1}) on the North slope was more than double that of reseeded grasses and nongraminoids on the South slope. For reseeded *P. maximum*, UPR had nearly two-fold higher biomass yield than UPR and CR in the South aspect. For *E. curvula*, biomass yield was the highest in CR on the South slope, being nearly two times higher than UPR and CR on the North slope and UPR on the South slope. The nongraminoid biomass in CO on the South-facing slope was more than double that of nongraminoids on the North aspect (Table 3).

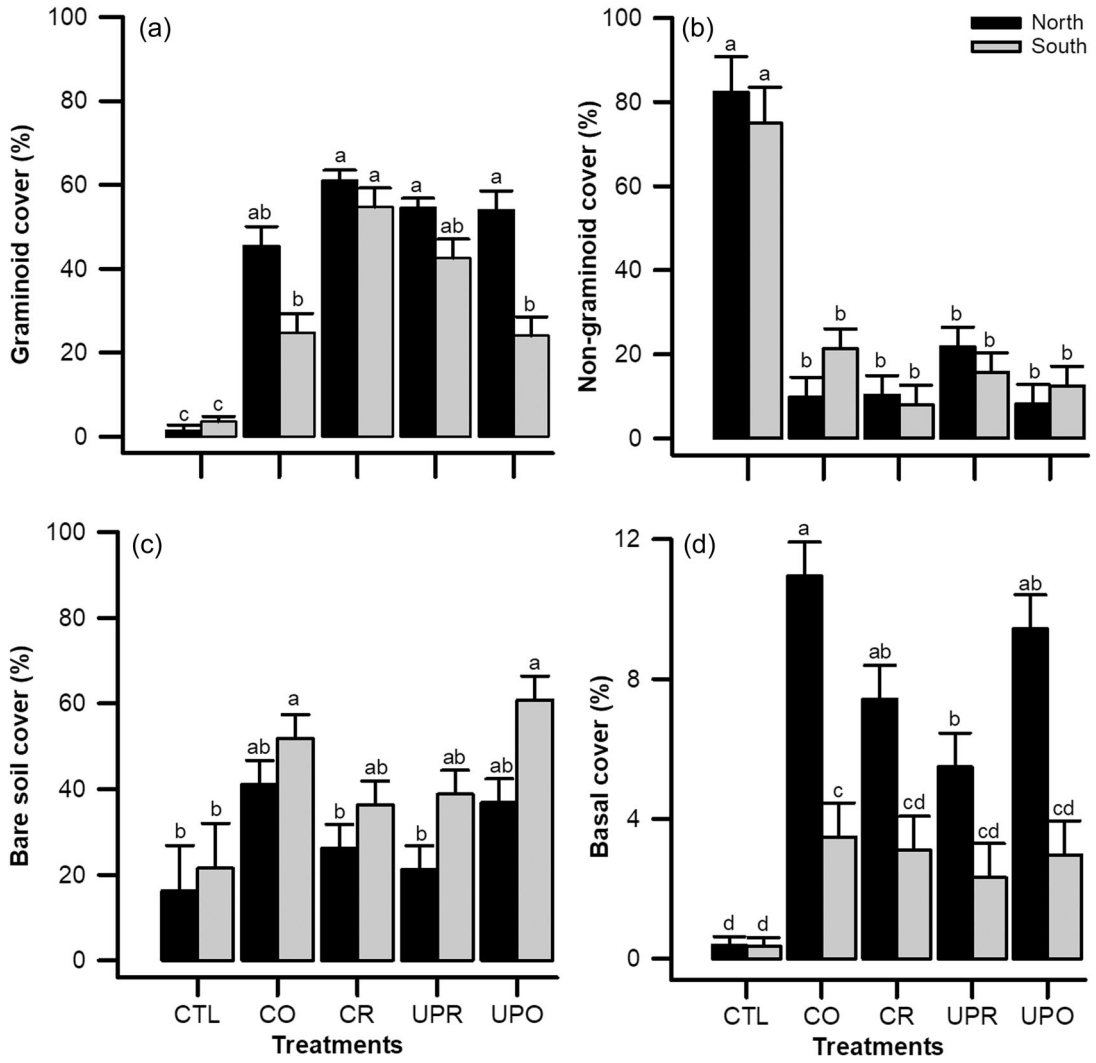


FIGURE 4 (a) Graminoid cover, (b) nongraminoid cover, (c) bare soil cover and (d) basal cover in various restoration treatments. Different letters on the graphs denote a significant difference in plant covers ($p \leq 0.05$). CO, cutting only; CR, cutting and reseeding; CTL, control treatment; UPO, uprooting only; UPR, uprooting and reseeding.

TABLE 3 Biomass production (kg DM ha^{-1}) of herbaceous plant species under various restoration treatments for North (N) and South (S) aspects.

		Restoration treatments				
Plant species	**Aspect**	**CTL**	**CO**	**CR**	**UPO**	**UPR**
Panicum maximum	N	0e	0e	380bc	0e	500bc
	S	0e	0e	160bcd	0e	260cd
Eragrostis curvula	N	0e	0e	580ab	0e	560ab
	S	0e	0e	860a	0e	400bc
Nonreseeded grasses	N	220cd	1820a	660ab	1400a	300bc
	S	100cd	740ab	280bc	680ab	540bc
Nongrass plants	N	20e	267c	200bcd	280abc	480bc
	S	80d	920a	380bc	720ab	540bc

Note: Different letters denote a significant difference between means ($p < 0.05$).
Abbreviations: CO, cutting only; CR, cutting and reseeding; CTL, control treatment; N, north; S, south; UPO, uprooting only; UPR, uprooting and reseeding.

Correlations between herbaceous vegetation attributes

Graminoid cover had a moderate positive correlation with BC ($r = 0.55$) and biomass production ($r = 0.53$). Nevertheless, graminoid cover had a fairly strong negative correlation with nongraminoid cover ($r = -0.69$). Nongraminoid cover had a very weak negative relationship with litter ($r = -0.21$) and biomass ($r = -0.25$), but a moderate negative correlation with BC ($r = -0.5$) and bare soil ($r = -0.57$). Bare soil cover had a weak negative relationship with plant biomass yield ($r = -0.28$, Table 4).

DISCUSSION

Effect of shrub clearing and restoration on diversity, richness and density

The various range rehabilitation techniques have been found to affect herbaceous plant density, species richness

TABLE 4 Correlation matrix between biomass production (BP), graminoid cover (GC), nongraminoid cover (NGC), litter cover (LC) and bare soil cover (BSC) on rehabilitated communal rangelands.

Item	BP	GC	NGC	LC	BSC
BP					
GC	**0.53*****				
NGC	**−0.25****	**−0.69*****			
LC	0.08NS	0.03NS	**−0.21***		
BSC	**−0.28****	−0.20NS	**−0.57*****	0.09NS	
BC	−0.03NS	**0.55*****	**−0.50*****	0.17NS	0.03NS

Note: Bold values indicate significant correlations between two variables.

Abbreviation: BC, basal cover.

***Denotes significance at $p < 0.0001$, **denotes significance at $p < 0.01$, *denotes significance at $p < 0.05$ and NS denotes non-significant correlations between two parameters at $p < 0.05$.

and diversity (Figure 3a). All restoration treatments increased species diversity and richness, highlighting that the encroachment of *P. incana* is detrimental to herbaceous vegetation (Figure 3b,c). The effects of the clearing only treatment were similar to other restoration treatments, indicating that high species diversity can be enhanced without the introduction of species through overseeding. This concurs with Mndela et al. (2023), who reported that increased light intensity after clearing *Seriphium plumosum* L. shrub-encroached areas increased the density, richness and diversity of herbaceous plants. The results of this study indicate that a release from shrub competition is enough to facilitate herbaceous recovery. Similarly, Abdellatif et al. (2016) and Mangani et al. (2022) found that reseeded sites did not vary from natural recovery in species richness and diversity.

In this study, it was noted that the UPO treatment, in particular, enhanced the regeneration of pioneer grasses, especially *Eragrostis* species and *A. congesta*, which had been displaced at encroached sites (Table 1), indicating that soil disturbance could have activated the seed bank of these species. These findings suggest that reseeding may be unnecessary for the rehabilitation of encroached sites. However, it is worth mentioning that the occurrence of pioneer species in UPO treatment means that long-term protection of these sites is needed following shrub clearing to allow successional vegetation changes from pioneer grasses to dominance of climax grasses. This process may take 10–20 years (Chazdon et al., 2021), meaning that land owners, particularly livestock farmers, need a reserve pasture for grazing. Otherwise, if grazing occurs at this stage, it may facilitate retrogression such that soils will be exposed to runoff and erosion. Thus, in instances where farmers want to utilise their rangelands very early, reseeding would be the option in either cut or uprooted shrub sites, as reseeded species performed better in these treatments (Table 1). Soil disturbance during uprooting tends to move deeply buried seeds to a sub-surface layer, where germination can be easily stimulated. Disturbed sites may also be safe sites for germination of seeds dispersed from neighbouring sites (Arruda et al., 2018; Cosyns et al., 2006). Reseeding would be more beneficial for increasing plant densities, as indicated by higher densities in UPR treatments, despite non-statistical differences from other restoration treatments (Figure 3a). This was ascribed to increased recruitment of reseeded species (*P. maximum* and *E. curvula*). Uprooting not only activates the seed bank of native species but also promotes seed burial, seed–soil contact and en masse recruitment of reseeded species (Stromberg et al., 2007). Conversely, in uncleared CTL, there is high shading and competition for below-ground resources by *P. incana*, leading to the reduction of densities and termination of shade-intolerant species in the system (Figure 3a; Kapiyana et al., 2025). Similarly, Davies et al. (2024) found higher plant density in reseeded plots in encroached areas.

Interaction of aspect and restoration techniques in plant canopy and BC

The restoration methods increased the graminoid and BC, but their effectiveness was dependent on the slope aspect. In the North aspect, all treatments increased the graminoid and BC relative to the CTL. This is not surprising, given that most native species, including reseeded species, are warm-season C_4 grasses, whose recruitment and production depend largely on high light intensity in the North slope. Thus, removal of the shrub not only released plants from below-ground competition but also released them from shrub canopy shading, which allowed native and reintroduced species to regenerate in high densities. In the South aspect, treatments UPR and CR increased only the graminoid cover (Figure 4a,d), while the other two treatments had no significant effects on BC and graminoid cover compared to the CTL. This is attributed to the South aspect being cooler than the North aspect due to reduced solar radiation (Gxasheka & Dlamini, 2024), which promoted the abundance of C_3 plants, which mostly were forbs (Table 1). This was further justified by the occurrence of a C_3 grass (*K. curva*) in South-facing slopes, indicating that site conditions were more favourable for cool-season plants.

However, in South-facing slopes, reseeding, either combined with cutting or uprooting, significantly increased graminoid cover relative to unreseeded restoration treatments. This highlights that either reseeded species have high

phenotypic plasticity to adapt to both North and South slopes or conditions in the South slope were within the acceptable range of these species. In reseeded plots, particularly *E. curvula*, together with other native species, formed dense homogeneous cover, highlighting that the species effectively outcompeted nongraminoids (Brown, 2023; Roberts et al., 2021). *E. curvula* is reported to be highly competitive against other plants due to its high seed production, germination rate and growth even in less fertile soils (Alagbo & Chauhan, 2023). Similarly, in the studies by Guevara et al. (2005) and Brits (2007), *E. curvula* attained a relatively higher cover in reseeded sites.

On the other hand, the low graminoid cover in unreseeded treatments on the South slope could be due to a low seed bank and or lack of propagule dispersal to shrub-cleared sites to facilitate passive recovery. These findings show that postclearing management of previously shrub-encroached areas in South slopes requires the introduction of target species through reseeding, as passive recovery proved to be ineffective. Reseeding of native grasses increases competition against exotic plants (Smith et al., 2021), thereby enhancing ecosystem resilience. Similarly, the meta-analysis by Monaco and Gunnell (2020) reported increased grass cover in reseeded compared to unreseeded sites. UPO was more effective in restoring the graminoid cover in the North aspect than in the South aspect, probably due to higher seed and propagule banks in the North compared to the South slope (Figure 4a). Uprooting disturbs the soil and enhances seed and propagule movement close to the surface, thereby enhancing seedling emergence of native grasses. This concurs with Li et al. (2021), who found higher cover of graminoids in the drier slopes relative to cooler slopes.

However, it is worth noting that, despite increased BC in restoration sites, BC was relatively low. These findings indicate that although restoration enhanced plant density, the vigour of these plants was poor, with relatively small grass tufts. This is ascribed to the variation in microclimate between the slope aspects, given that most grasses here are C_4 and the South-facing slope favours C_3 plants. Angassa et al. (2010) found a decrease in graminoid cover as BC declined. These findings align with previous studies of Abdeta et al. (2012) and Mndela, Madakadze, Nherera-Chokuda, et al. (2022), which reported that SE decreases the BC of herbaceous plants.

All restoration treatments decreased the nongraminoid cover compared to the CTL, which showed higher nongraminoid cover (Figure 4b). This is due to the variation in light intensity between restoration treatments and the CTL. The correlation matrix indicated that an increase in graminoid cover negatively correlated with nongraminoid cover, suggesting that grasses outcompete or replace forbs after shrub clearing. Moreover, CR and UPR were the only restoration treatments that effectively reduced bare patches in the North aspect, probably due to higher recruitment of reseeded species that covered more ground than in cleared and UPO treatments. This aligns with Tebeje et al. (2014), who found higher vegetative cover and less bare soil in reseeded than cleared and nonreseeded sites.

Response of herbaceous plant biomass to restoration of shrub-encroached rangelands

The restoration techniques improved biomass production, but the degree of improvement depended on the slope aspect and species (Table 3). Nonreseeded grasses had the highest biomass yield in CO in the North aspect, while reseeded grasses and nongraminoids had the highest biomass yield in CO in the South aspect. However, the nongraminoid biomass was double in the South aspect compared to the North-facing slope (Table 3). These findings are ascribed to variations in microclimate between the slope aspects, with the North aspect facilitating the recovery of C_4 grass more than forbs, whereas the South aspect promotes the abundance of C_3 grasses and nongraminoids. Furthermore, this is also justified by the high abundance of nongraminoids in the South-facing slope (Table 1). Although the biomass of reseeded *E. curvula* did not differ significantly between the restoration treatments on the two slope aspects, CR in the South aspect had the highest biomass (Table 3). This suggests that *E. curvula* has the ability to thrive also in South-facing slope sites and produce high biomass yield. These findings concur with those studies that reported that *E. curvula* can adapt to different ecological conditions with high biomass production (Gherbin et al., 2007; Motsomane & Magadlela, 2025). Shrub clearing and grass reseeding restoration techniques effectively increased the grass cover relative to unreseeded sites, and this could be attributed to low seed bank status. However, reseeding with *E. curvula* in the South aspect will be beneficial to local farmers by increasing forage availability compared to unreseeded sites.

CONCLUSIONS

The results of the study showed that the restoration treatments promote herbaceous vegetation restoration, with integration of shrub clearing and reseeding being more effective in fast-tracking restoration process in the semi-arid rangelands of South Africa. However, the responses of herbaceous cover and biomass to restoration treatments varied by slope aspect, signifying that the geographical details of the site need to be considered when designing a restoration management plan. The results of the study form the basis for designing management of SE and for restoration of herbaceous vegetation. The results of this study can be used for development of policy associated with vegetation conservation and restoration. However, the short-term nature of this study limits the understanding of the long-term vegetation dynamics in restoration sites. Hence, future long-term monitoring of vegetation changes would provide more insights into successional community dynamics and further guide proper shrub management and eradication. We also note that this study was conducted at a small plot level, thus limiting large landscape-scale monitoring of other ecosystem services, for example, carbon sequestration. Therefore, further future research is merited to monitor the long-term effects of restoration on long-term vegetation

changes and their associated effects on carbon sequestration and water–energy balance at a large landscape scale. Such research can incorporate the use of remote sensing and machine learning to determine the impact of changes in vegetation over time in restoration sites.

AUTHOR CONTRIBUTIONS

Monelisi Makanya: Conceptualization; data curation; formal analysis; investigation; methodology; visualization; writing—original draft; plant taxonomy. **Khanyisa Jama:** Conceptualization; funding acquisition; investigation; methodology; project administration; resources; supervision; validation; visualization; writing—review and editing. **Mthunzi Mndela:** Conceptualization; data curation; formal analysis; investigation; methodology; resources; software; supervision; validation; visualization; writing—review and editing; plant taxonomy. **Sinalo Malindie:** Conceptualization; investigation; methodology; project administration; resources; validation; visualization; writing—review and editing.

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CONFLICT OF INTEREST STATEMENT

The authors declare no conflicts of interest.

DATA AVAILABILITY STATEMENT

The data used in this study will be made available on request.

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