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RESEARCH ARTICLE

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Estimating mountainous plant species richness and diversity for monitoring global change in a protected grassland park

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Abstract

Assessments of species diversity and richness are essential to understand present ecological and biodiversity conditions for effective conservation management strategies. Biodiversity indicators determine rangeland health and response to grazing, fire regimes and climate change. This research examined species richness, diversity and composition in a protected mountainous grassland. Two data sets, both collected from a 30 × 30 m plot, with similar species composition and cover were combined. One data set was collected using a 100-step point survey and the other from a series of 16 plots. A single-factor analysis of variance was used to test if the mean species richness and diversity of the sites differed across the study area. Species accumulation curves were used to determine the relationship between species richness and the number of sampling units per site. The results from fitting a species–area equation showed that the estimated maximum species richness was slightly greater than the observed species pool in all sites, meaning that the sampling units were not adequate (albeit by small margins) to capture all vascular plant species in the sites. Diversity metrics could, thus, be used to monitor species change within grassland plant communities.

KEYWORDS

afromontane grasslands, community ecology, conservation, GIS, mesic grasslands, plant diversity

Résumé

Les évaluations de la diversité et de la richesse des espèces sont essentielles pour comprendre les conditions actuelles de l'écologie et de la biodiversité afin de mettre en place des stratégies efficaces de gestion de la conservation. Les indicateurs de biodiversité déterminent la santé des parcours et leur réaction au pâturage, aux régimes d'incendie et au changement climatique. Cette étude a porté sur la richesse, la diversité et la composition des espèces dans une prairie de montagne protégée. Deux ensembles de données, tous deux collectés sur une parcelle de 30 × 30 m, présentant une composition d'espèces et une couverture similaires, ont été combinés. Un ensemble de données a été collecté à l'aide d'une enquête ponctuelle en 100 étapes et l'autre à partir d'une série de 16 placettes. Une analyse de variance

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à un seul facteur a été utilisée pour vérifier si la richesse et la diversité moyennes des espèces des sites différaient dans la zone d'étude. Des courbes d'accumulation d'espèces ont été utilisées pour déterminer la relation entre la richesse en espèces et le nombre d'unités d'échantillonnage par site. Les résultats de l'ajustement d'une équation espèce-surface ont montré que la richesse maximale estimée des espèces était légèrement supérieure à la réserve d'espèces observée dans tous les sites, ce qui signifie que les unités d'échantillonnage n'étaient pas suffisantes (bien que par de faibles marges) pour capturer toutes les espèces de plantes vasculaires dans les sites. Les mesures de la diversité pourraient donc être utilisées pour suivre l'évolution des espèces au sein des communautés végétales des prairies.

1 | INTRODUCTION

Biodiversity at the species level is deteriorating rapidly due to a range of human activities resulting in global environmental changes (GEC; Hooper et al., 2012; Steffen et al., 2004). The main drivers of GEC – increasing atmospheric CO_2 levels and associated climate change, excess deposition of nitrogen, loss and fragmentation of natural habitats and biotic invasion – may result in extinction and alteration of species distribution (Hooper et al., 2012; Steffen et al., 2004). Nearly 50% of terrestrial ecosystems have been transformed by direct human actions, with negative consequences for biodiversity, nutrient cycling, soil structure, soil organisms and climate (Steffen et al., 2004). Studies show that the impact of species loss is comparable to the impact of GEC (Cardinale et al., 2018; Hooper et al., 2012). Plant biodiversity loss may reduce plant production and alter nutrient cycling (Balvanera et al., 2006; Cardinale et al., 2011). Hence, Hooper et al. (2012) argue that biodiversity loss in the 21st century could be one of the major drivers of ecosystem change.

High levels of biodiversity are essential for the long-term resilience of ecosystem functions, processes and services (Oliver et al., 2015). All species in a community influence ecosystem properties and services; however, this is dependent on their respective functional traits (Díaz et al., 2013). Equally, ecosystem changes affect species' functional traits, i.e., morphological, biochemical, physiological, structural, phonological or behavioural characteristics, thus altering ecosystem functions and resilience (Díaz et al., 2013; Suding et al., 2008). Therefore, biodiversity loss may reduce the efficiency of plant communities in capturing biologically essential resources such as water, nutrients and sunlight, which negatively influence organic nutrient recycling via decomposition (Cardinale et al., 2012). Biodiversity–ecosystem functioning relationships are affected by the number and identities of species, their evenness, functional traits and interactions within the community.

Organisms can influence the physical formation of habitats and element concentrations in biogeochemical cycles via ecological engineering and stoichiometry, respectively (Glibert, 2016; Jones et al., 1994). Changes in an ecosystem are attributed to the physiology, morphology and behaviour of individual organisms at a species level and structure and composition at the population and community levels (Suding et al., 2008). In grassland communities, increased plant species richness increases plant productivity and nitrogen input (Tilman et al., 2012). Therefore, assessing the levels of species diversity assists in understanding resource use by and distribution of grazing animals. In addition, understanding the distribution of mountainous species diversity is essential for identifying priority areas for conservation (Barros et al., 2015). Research shows that high plant diversity often increases plant productivity, nutrient cycling and ecosystem stability in grassland plant communities (Tilman, 1999; Tilman & Downing, 1994). The distribution of large grazing mammals is attributed to the occurrence of nutritionally enriched vegetation species in Serengeti National Park, Tanzania (McNaughton, 1988). Good baseline data that assess biodiversity are essential for monitoring and predicting future biodiversity patterns, which will immensely improve conservation to abate further loss (Cardinale et al., 2018).

Most losses of biodiversity are observed at a global scale where both increases and decreases in community diversity are recorded at regional and local scales (Reitalu et al., 2012). Global environmental change affects species' tolerances and habitats. Therefore, a species with restricted ecological niches is more susceptible to extinction than those with broader ecological tolerance occupying more habitats (Reitalu et al., 2012). Improved data sets explaining patterns of biodiversity in an ecosystem will assist in understanding the current and probable future consequences of global environmental changes (Wardle et al., 2011). Exploring data sets that could explain plant diversity and ecosystem relationships improves our understanding of how biodiversity loss impacts natural communities (Wardle, 2016). Measures of biodiversity parameters are important indicators of ecosystem health in rangelands. Species richness and diversity are two metrics that are useful and appropriate because they are easy to measure and interpret (Symstad & Jonas, 2011). In addition, useful information about metrics of diversity is derived from species–area equations (Malanson et al., 2020). Assessment of plant diversity data, especially about species occurrence and abundance, reveals the concentration of dominant species, thus enhancing efforts to save biodiversity worldwide (Gaury & Devi, 2017). Hence, within protected areas, data on species richness and diversity are important for ecological explanations, which enable designs of scientifically defendable conservation strategies (Brown et al., 2013).

Mountain ecosystems in Southern Africa have high levels of biodiversity and species richness with complex ecosystems (Brown & du

Preez, 2020). Golden Gate Highlands National Park (GGHNP) forms part of the Maluti–Drakensberg Mountain range and is a grassland biome that contains a variety of aesthetically pleasing landscapes with high biodiversity value (Kay et al., 1993). The park is extremely variable in topography and climate, which results in a complex mosaic of plant communities (Kay et al., 1993). Despite being home to diverse species and complex ecosystems, these grasslands are prone to human-induced pressure such as climate change and biological invasions. The lack of data and protection of these biomes exacerbate the vulnerability to and risk of extinction. Due to the accessibility of GGHNP to the public and local communities, the park's resources are exploited. Overgrazing, thatch harvesting, overstocking and unplanned fires negatively affect the biodiversity of the park. As part of developing species inventories for long-term monitoring, the South African National Parks collects vegetation data in GGHNP. The data are used for vegetation classification and taxonomic species composition (Kay et al., 1993), and land degradation indicators. Unfortunately, data on species diversity and richness for many parts of the grassland biome in Southern Africa are not always available and seldom analysed to determine species diversity and richness, especially in sub-alpine grassland communities of the Drakensberg (Brown & du Preez, 2020). Available information indicates that the grassland biome of Southern Africa contains a mean plant richness of 82 species per 1000 m^2 (Brown & Bezuidenhout, 2020; Eckhardt et al., 1996), and between 9 and 29 species have been recorded in 100 m^2 vegetation sampling plots within high altitude grassland of the Eastern Cape and KwaZulu Natal provinces in South Africa (Eckhardt et al., 1996). Thus, the grassland biome is considered the most diverse with alpha diversity in South Africa, second to the fynbos (Brown & Bezuidenhout, 2020). Quantitative estimates of the levels of biodiversity are important for improving our understanding of subsequent changes in ecosystem functioning (Cardinale et al., 2012). This research assessed and explored baseline data sets to test hypotheses about changes in species richness to answer the following research questions: (1) which sites are habitat to a high number of plant species in GGHNP? (2) Are there any significant variations of species richness and diversity across sites, fire severity and grazing pressures in the park? and (3) What is the estimated number of species within each site?

2 | METHODOLOGY

2.1 | Study area

The study was conducted at GGHNP, which is in the North-Eastern parts of the Free State province, South Africa (Figure 1). The park covers 32,758.35 ha and lies between 28°27′S–28°37′S and 28°33′E–28°42′E. The GGHNP is a mountainous grassland located at the foothills of the Drakensberg and forms part of the mesic highveld grassland with marked variation in geology, topography and rainfall. The following soil types were identified in the park: shallow rocky soils (Glenrosa and Mispah), deep soil along drainage lines (Oakleaf), well-developed sand soils, e.g., Hutton and Clovelly, as well as Clayey structured soils including Milkwood and Tambakulu (SANParks, 2020). The park is characterised by summer rainfall, temperate summers and cold winter. The rainfall season stretches from September to April, with a mean annual rainfall ranging from 800 to 2000 mm (Kay et al., 1993). The park lies between 1892 and 2829 m above sea level and comprises the following grasslands units: Eastern Free State sandy grasslands (Gm 4), Basotho montane shrubland (Gm 5), Lesotho highveld basalt grassland (Gd 8) and Northern Drakensberg highveld (Gd5) (Mucina & Rutherford, 2006).

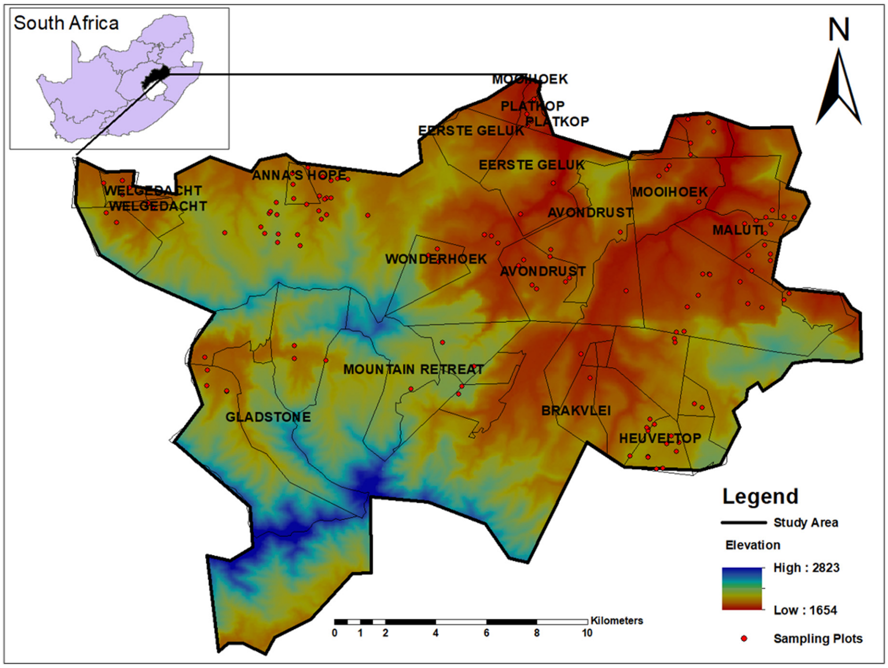

FIGURE 1 A map of the study area.

2.2 | Sampling design and data collection

The land type map of GGHNP was used as the first stratification. Sampling sites of relatively homogenous grass units were then located in a randomly stratified manner. A total of 36 vegetation sampling plots (30×30m) ranging between three and five per site (six sites with five plots and two sites with three plots; Figure 2a) were placed randomly within the homogenous grass units. Within each sampling plot, a total of 16 (1×1m) quadrats were placed systematically at every 10m along four parallel rows (Figure 2). The above-mentioned data set was merged with another data set from the same research comprising 12 sites with 106 plots, collected using 100-step points from four transects located within 30×30m plot where all species were recorded at every step point (Figure 2b). The combined data set comprised 142 plots with a total of 13 sites, seven of which were the same and six different (Table 1). The distribution of sampling plots across the land types in GGHNP is given in Figure 3.

In each quadrat, all vascular plant species of the standing vegetation were identified to species level where possible and the visually estimated aerial cover of each species was estimated to the nearest 5% with the quadrat. In each sampling plot where step-point transects were done, Braun-Blanquet cover-abundance values were used as a proxy for species cover.

The taxonomic composition and percentage cover of both visually estimated quadratic based and Braun-Blanquet abundance values of each vascular plant species were used to derive species richness and subsequently diversity per plot (Tilman et al., 2006). Subsequently, the values from each plot were averaged to attain mean species richness per site. The variables species richness and diversity were computed using "vegan" and "plyr" R statistical packages (Liang & Seyfried, 2001; Oksanen, 2017), which employed the diversity and apply function for species diversity and richness respectively.

Total species diversity was calculated using the Shannon–Wiener Index (Equation 1) where pi is the proportion of the species within the sampling units. Species richness was determined by adding all species from each quadrat and averaging by the number of quadrats in each plot, to obtain the average plot value. Data on species richness and diversity across selected sites are given in Table 2.

$$H' = -\sum pi^{*}\ln pi \tag{1}$$

2.3 | Fire data collection

A burn severity map was created for areas affected by fires. The normalised burn ratio (NBR), which utilises near-infrared (NIR) and shortwave-infrared (SWIR) wavelengths, were used because it highlights burned areas and estimates burn severity (Hawbaker et al., 2020). The NIR and SWIR bands were obtained from Landsat 8 satellite imagery available on the Google Earth Engine. These bands

TABLE 1 The number of plots at each site from quadrat and step-point vegetation plot surveys.

Site name	Quadrat-based plots	Step-point survey plots	Merged data set
Annas' hope	5	23	28
Avondrust	–	9	9
Basotho	3	18	21
Brakvlei	3	2	5
Eerste Geluk	–	5	5
Gladstone	5	3	8
Heuvel Top	5	16	21
Maluti	5	6	11
Mooihoek	5	5	10
Mountain Retreat	5	–	5
Platkop	–	5	5
Welgedacht	–	8	8
Wonderhoek	–	6	6
Total	36	106	142

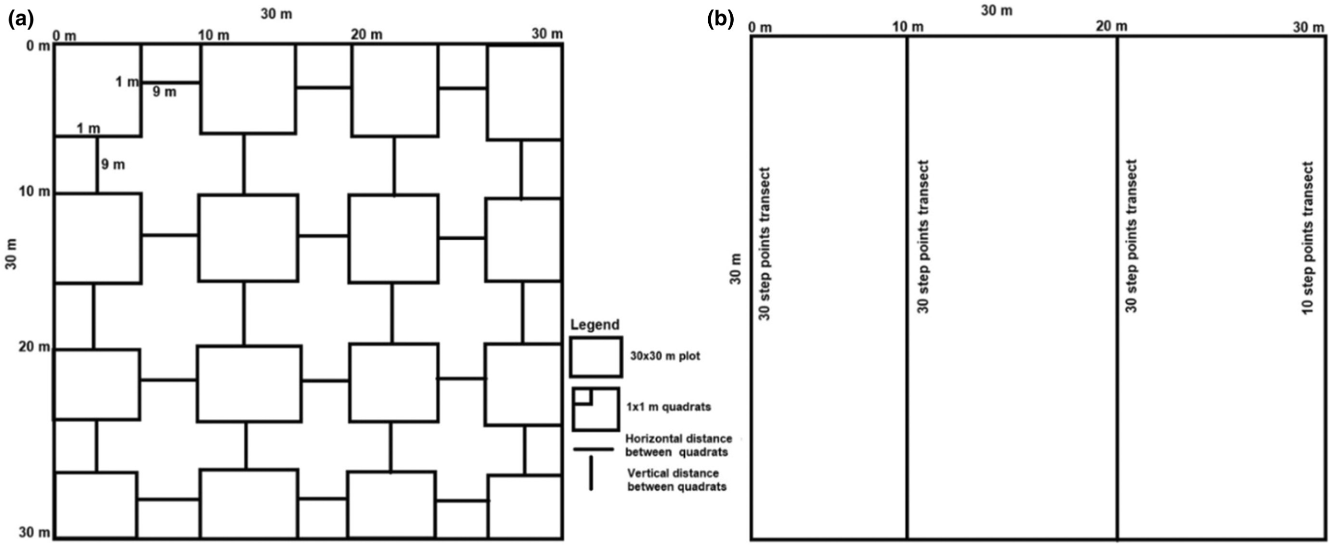

FIGURE 2 (a) Quadrat and (b) step-point-based vegetation plot design.

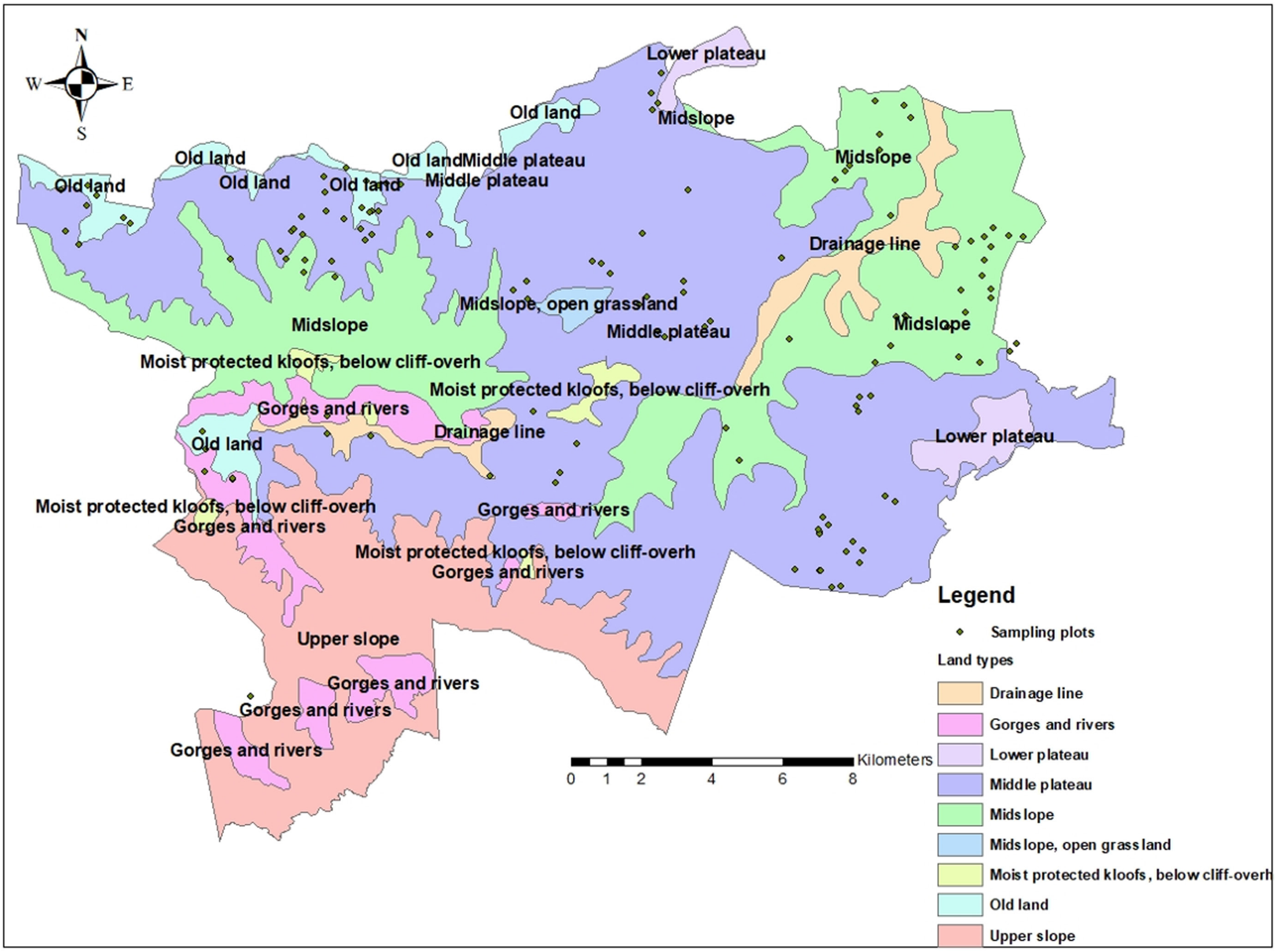

FIGURE 3 Distribution of sampling plots across the land types in Golden Gate Highlands National Park.

were used to calculate the NBR for pre-and post-fire scenarios. The bands were selected from images of the year before the vegetation survey (2018), before the start of the fire season (January–April), and post-fire season (June–September). Subsequently, an image of delta NBR was determined through the differences between pre-and post-fire NBR (Equations 2 and 3). Finally, the ΔNBR was classified according to the United States Geological Survey (n.d.; Table 3).

$$\mathrm{NBR} = \mathrm{NIR} - \mathrm{SWIR} / \mathrm{NIR} + \mathrm{SWIR} \tag{2}$$

$$\Delta\mathrm{NBR} = \mathrm{NBRprefire} - \mathrm{NBRpostfire} \tag{3}$$

2.4 | Grazing data collection

Animal census data of grazers collected in the park during 2020 were used for the determination of grazing pressure in the park. The data are collected biennially by surveying the area from a helicopter and counting the number of animals including herd sizes found in the park. The hotspot analysis tool Getis–Ord Gi statistics, which identifies statistically significant hotspots and cold spots, was used to find where animals cluster spatially within the park (Getis & Ord, 2010). Subsequently, inverse distance interpolation was used to estimate cell values with the highest and lowest animal concentrations. Estimated values ranging from −1 to 1 were regarded as cold spots and therefore classified as low grazing intensity while 1–3 values were regarded as hotspots and classified as high grazing intensity.

2.5 | Data analysis

The merged data set was used to compute a single-factor analysis of variance (one-way ANOVA) which was used to test if the mean species richness and diversity of sites, fire severity, and grazing intensity differed across the study area. Subsequently, Tukey's honest significant difference was used to test the significance of inter-site variations in richness and diversity. Statistical analyses were performed using R-software and a significant difference was considered at a 95% confidence level ($p < 0.05$).

The data set from the 36 plots comprising eight sites was used to compute species accumulation curves (SACs) to determine the adequacy of the quadrat-based vegetation sampling plots in capturing local species richness and diversity of sites. A non-linear regression (nls) model for species–area was fitted to the SACs for attaining diversity parameters where the asymptotic maximum

TABLE 2 Data summary of species richness and diversity (mean ± standard deviation and coefficient of variability in percentages [CV]) and plant species richness across selected sites (*n* = the number of plots per site).

Site name (*n*)	Grasses' richness	Forbs' richness	Shrubs' richness	Total number of species	Total species richness (mean ± SD)	CV	Total species diversity (mean ± SD)	CV
Welgedacht (8)	17	10	0	27	11.63 ± 2.26	19	1.90 ± 0.31	16
Annas' hope (28)	41	13	4	58	10.92 ± 3.39	31	1.83 ± 0.35	21
Wonderhoek (10)	19	7	0	26	10.67 ± 1.75	3	1.78 ± 0.36	20
Avondrust (6)	19	9	1	29	9.78 ± 4.12	42	1.76 ± 0.28	31
Mooihoek (10)	17	9	3	29	9.52 ± 3.28	40	1.74 ± 0.38	20
Maluti (11)	23	7	0	39	9.26 ± 3.74	40	1.71 ± 0.24	19
Eerste Geluk (5)	7	10	3	29	8.84 ± 2.64	30	1.67 ± 0.28	14
Brakvlei (5)	9	16	1	26	8.25 ± 3.97	48	1.67 ± 0.50	21
Basotho (21)	23	7	1	31	8.21 ± 3.62	44	1.55 ± 0.49	32
Gladstone (8)	26	23	3	52	7.80 ± 1.73	22	1.48 ± 0.34	15
Heuvel Top (21)	23	7	2	30	7.37 ± 2.95	50	1.45 ± 0.31	23
Platkop (5)	17	8	2	27	5.89 ± 1.43	15	1.16 ± 0.18	36
Mountain Retreat (5)	19	9	0	28	4.28 ± 0.57	13	1.15 ± 0.41	16

TABLE 3 Burn severity levels.

Burned severity class	ΔNBR
Unburned	−0.10 to 0.10
Low severity	0.10 to 0.27
Moderate–low severity	0.27 to 0.44
Moderate–high severity	0.44 to 0.66
High severity	>0.66

number of species was considered the optimal sampling plot for determining species richness. The Lomolino model was the nls: Asym/$(1+\text{slope}^{\log(\text{xmid}/\text{area})})$, where parameter Asym is the asymptotic maximum number of species, the slope is the maximum slope of increase of richness, and xmid is the area, where half of the maximum richness is achieved (Lomolino, 2000). Gamma diversity, which is the highest number of species in each site, was derived from the "specaccum" function found in the vegan package of R statistical software (Equation 4; Oksanen, 2017).

$$S = S_{\max} / \left(1 + \text{slope}^{\log(\text{xmid}/\text{area})}\right) \tag{4}$$

3 | RESULTS

There is a significant difference in species richness between sites in the park ($F = 3.55$, $p < 0.05$). Similarly, species diversity is also significantly variable across the park ($F = 2.79$, $p < 0.05$; Table 4). However, species richness and diversity did not differ significantly between burn severity classes ($F_{\text{Richness}} = 1.81$, $p_{\text{Richness}} = 0.13$, $D_{\text{iversity}} = 0.45$, $p_{\text{Diversity}} = 0.77$) and grazing pressures ($F_{\text{Diversity}} = 0.67$, $p_{\text{Diversity}} = 0.42$, $F_{\text{Richness}} = 3.42$, $p_{\text{Richness}} = 0.06$; Table 4). The Tukey HSD showed that 61% of sites showed no significant variation, while only 39% of the sites exhibited significant variation in species richness and diversity.

The plots at Annas' hope and Welgedacht showed significantly ($p < 0.05$) higher species richness compared to Mountain Retreat and Heuvel Top. Sites such as Avondrust, Basotho, Brakvlei, Eerste Geluk, Gladstone and Platkop showed similar richness values and were not significantly different ($p > 0.05$). Plots at Welgedacht showed a significantly higher species diversity than the rest of the sites ($p < 0.05$). Platkop and Mountain Retreat were significantly lower than the rest of the sites ($p < 0.05$). In general, the remaining sites did not differ in species diversity ($p > 0.05$).

Species diversity was similar between fire severity classes. Species richness did not differ between fire severity classes ($p > 0.05$). However, unburned sites had the highest species richness while sites with moderate to high severity had the lowest species richness. Similarly, both species diversity and richness did not differ between grazing pressures (Table 4).

The fitted Lomolino model estimated that the asymptotic number of species in a site was less than the gamma diversity (species pool was the total number of species in each site). The slope of the species accumulation curve ranged between 2.07 and 2.35 (Table 5). In addition, the species accumulation curves showed that our sites were approaching asymptote of the maximum number of species, albeit not reached.

4 | DISCUSSION

The sampled sites in GGHNP are habitat to a high number of plant species representing a high gamma diversity in the grasslands, and therefore a crucial component of the biome. This is consistent with the high-altitude grasslands of KwaZulu Natal and Eastern Cape provinces where the species ranged between 9 and 29 from 100 m^2 vegetation sampling plots (Eckhardt et al., 1996). Patterns of species richness among plant communities vary through space and time in response to endogenous and exogenous factors (Bagchi et al., 2017). Furthermore, fire and grazing may interactively or non-interactively promote heterogeneity in the composition and diversity of grasslands (Symstad & Jonas, 2011). Our results indicate a significant variation of species richness and diversity across the park, even though

TABLE 4 Descriptive statistics of species richness and diversity (*H′*) and ANOVA at park level.

Variables of interest	Min	Max	Median	Mean	STDEV	CV	*F*-value	*p*-Value
Species richness (sites)	2.5	17	9	8.97	3.49	39.9	3.55	0.01
Species diversity (sites)	2.5	17	9	8.97	3.49	24.63	2.79	0.01
Species richness (fire severity)	2.5	17	9	8.97	3.49	39.9	1.81	0.13
Species diversity (fire severity)	2.5	17	9	8.97	3.49	24.63	0.45	0.77
Species richness (grazing pressure)	2.5	17	9	8.97	3.49	39.9	3.42	0.05
Species diversity (grazing pressure)	2.5	17	9	8.97	3.49	24.63	0.67	0.42

TABLE 5 The Lomolino model parameters for the fitted species accumulation curves.

Site name (*n*)	Asymptote	Xmid	Slope	Gamma
Annas' hope (5)	56.96	10.49	2.12	49
Basotho (3)	39.94	8.52	2.12	32
Brakvlei (3)	50.1	8.73	2.6	42
Gladstone (5)	54.44	7.89	2.15	47
Heuvel Top (5)	35.72	10.95	2.35	30
Maluti (5)	58.12	10	2.5	51
Mooihoek (5)	42.27	9.71	2.28	36
Mountain Retreat (5)	42.5	15.87	2.07	33

most sites were similar in this regard. The attributes of the number of species at each site, variation or lack of species richness across sites might be an indication of environmental heterogeneity at different spatial scales (Filibeck et al., 2019) which is pivotal for promoting or maintaining plant richness. Moreover, the results emphasise the importance of spatially explicit data in assessing species richness and diversity within grassland terrestrial landscapes.

Grassland plant communities are highly dynamic due to both large- and small-scale disturbances (Collins et al., 2002). Hence, disturbances such as grazing and fire are pivotal for maintaining species richness in stable grassland (Poschlod et al., 1998). In this study, species richness and diversity did not differ amongst fire severity classes and grazing pressures. Sometimes grassland ecosystems show less response to grazing (Symstad & Jonas, 2011) and the effect of fire is highly variable but generally neutral to negative (Poschlod et al., 1998). Although species richness and diversity were significantly variable in the GGHNP, most inter-site variations were not significant. Such variations or lack thereof, are mostly attributed to the Intermediate Disturbance Hypothesis, which predicts that diversity will be maximised at intermediate levels of disturbance and lowered by extreme and scant disturbances (Brown et al., 2001). Therefore, species richness is a function of biotic and abiotic factors due to a specific set of species being attributed to a certain condition. Tolerance to environmental changes resulting from disturbances is also pivotal for regulating species richness in ecosystems (Brown et al., 2001). However, in some cases, species richness response to disturbances may be different from that of species diversity where the high number of species in a community does not translate to high species diversity.

The species accumulation curves in this study appear to be almost in the asymptote and the fitted Lomolino model estimated a slightly higher number of species richness in each site than the actual number recorded. For example, Annas' hope yielded a prediction of 57 species, while 49 species were recorded for the site. Similarly, Miller and Wiegert (1989) found that data sets of vascular plants have approached an asymptote albeit not necessarily reached in the biogeographic mountainous of Southern Appalachians. In ecosystems that are very difficult to sample, it is important to estimate the total number of species and the sampling effort needed to obtain these estimates (Ugland et al., 2016). Lomolino (2000) states that the relationship between species and sampling effort asymptotically approaches the maximum value of the species pool. Williamson et al. (2001) argue that this is only the case for a finite area with finite species.

The species accumulation curves in this for our quadrat-based data set were similar both in shape and completeness. SACs take account of species identities and the rate at which new species are encountered (Ugland et al., 2016). The shape of the SACs reflects on the evenness, abundances and distribution of species (Storch, 2016). In addition, asymptotic species richness estimates are not effective when the data set includes many rare species. Hence, Soberon and Jorge (1993) highlighted that near the asymptote rare species are likely to be the ones added. This may be the case in grasslands where grasses are the common and predominant taxa. Analysed separately, both rare and dominant species are likely to yield different estimates of species in GGNHP. Estimating the scale, i.e., temporal or spatial required to add a given number of species should make it possible to plan fieldwork in a rigorous manner to cut cost; therefore, SACs may be useful in GGHNP for monitoring and assessing species response to GEC (Flather, 1996; Malanson et al., 2020).

The fitted Lomolino model indicated that the slope of the accumulation curves ranged from 2.07 to 2.35. The slope of the plots is useful for comparing the degree of species turnover between different ecological sites (Filibeck et al., 2019). The slope indicates the rate of increase in diversity and therefore captures the beta diversity (i.e. variation of identities of species between plots of the observed scale [Malanson et al., 2020]). A higher slope of SACs implies a high spatial heterogeneity of vegetation. Therefore, the slopes of the fitted SACs in GGHNP can be used to determine sites with high species turnover, which are essential for setting conservation targets. Hence, monitoring diversity metrics can assist park conservation managers, especially for better response to biodiversity loss.

According to Symstad and Jonas (2011), "species accumulation curves are often recommended as an ideal indicator of biodiversity because they minimize complications of comparison across studies with varying sampling units". The key to the major distinction of species accumulation curves is that they measure the rate at which new species are found with increasing sampling effort (species turnover). In this study, the number of species inhabiting those sampling units increases; rapidly at first, but then more slowly for the larger sampling units, because the SACs were approaching an asymptote. Therefore, within our sites, species only became identical with a higher sampling size as indicated by the species accumulation curve approaching asymptote. Species richness can either increase, decrease or become regulated in space and time (Brown et al., 2001). This is mainly because environmental changes favour or disfavour some species, altering turnover in species composition (Brown et al., 2001). Indeed, the SACs will have more gradual slopes in heterogeneous areas than in homogenous areas. Knowledge of how spatial heterogeneity affects species pools is necessary for predicting species richness and turnover and therefore research on mechanisms controlling local species richness at different spatial scales is warranted.

5 | CONCLUSION

Ecological consequences of species loss are well studied (Hooper et al., 2012; Turnbull et al., 2016; Wardle, 2016). Thus, high levels of species richness and/or diversity are pivotal for maintaining many ecosystem functions and vice versa (Grman et al., 2018). Monitoring aspects of biodiversity richness and diversity as well as species composition is critical for managing ecosystem functions and services that benefit humanity and determines game-carrying capacity and stocking rates across landscapes. Our study explores data sets and ways in which species richness and diversity can be monitored across and between plant communities using species accumulation curves. Measurements of changes in local diversity (alpha) and species turnover (beta diversity) have been inadequate for average sites (Cardinale et al., 2018). This study provides a tool to monitor species change within grassland plant communities, and the information can then be used to manage and improve stocking rates and fire estimation models in the park.

CONFLICT OF INTEREST STATEMENT

No conflicts of Interest declared.

DATA AVAILABILITY STATEMENT

The data supporting this study's findings are available from the corresponding author upon request at 2018711909@ufs4life.ac.za.

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