The effects of fire-breaks on plant diversity and species composition in the grasslands of the Loskop Dam Nature Reserve, South Africa

Laura M Bachinger¹, Leslie R Brown² and Margaretha W van Rooyen*¹

¹ Department of Plant Science, University of Pretoria, Pretoria, South Africa
² Applied Behavioural Ecology and Ecosystem Research Unit, Department of Environmental Sciences, University of South Africa, Johannesburg, South Africa
* Corresponding author, email: Gretel.vanrooyen@up.ac.za

There is a dearth of knowledge on the effects of annual burning of fire-breaks on species composition, plant diversity and soil properties. Whittaker’s plant diversity technique was used to gather data on species composition and diversity in four grassland communities on the Loskop Dam Nature Reserve (LDNR). The study demonstrated that fire-breaks did not have a negative effect on plant diversity and an increase was even noted in various diversity parameters in the grassland on abandoned cropland. Fire-breaks were changing the species composition, as three of the four communities illustrated a clear separation in species composition between fire-break and unburnt plots. There was not a strong association between specific species and the fire-break or unburnt plots, except for Themeda triandra and Tristachya leucothrix, known for their association with and without fire, respectively. The change in species composition was not negatively affecting range condition. Carbon and nitrogen concentrations were slightly lower in the fire-break than unburnt soils, but the reduced concentrations were unlikely to cause severe soil degradation in fire-break zones. From a management perspective, fire-breaks appear to be a sustainable management tool as they are not adversely affecting plant diversity or range condition in the grassland association on LDNR.

Keywords: abandoned cultivated land, annual burning, range condition, species richness, succession

Online supplementary information: Supporting information for this paper is available as online supplementary material at http://dx.doi.org/10.2989/10220119.2015.1088574

Introduction

The Grassland Biome is the second largest of South Africa’s biomes and covers nearly one-third of the country’s land surface area (Mucina and Rutherford 2006). However, large proportions of this biome have been permanently transformed (Fairbanks et al. 2000; Mucina and Rutherford 2006) and it is therefore imperative that we manage the remaining grassland in a manner ensuring persistence of plant diversity. Intact grasslands support a high diversity of plant and animal species (O’Connor and Bredenkamp 1997). Within nature reserves these grasslands are important from an animal production and plant species diversity point of view. Not only do they support a large number of herbivores, but also a multitude of insect species whilst playing an important role in nutrient cycling and water retention (O’Connor and Bredenkamp 1997). Grassland vegetation generally occurs on flat, hilly or montane areas of South Africa and characteristically lacks a woody layer. However, grasslands also occur as open patches, often on plateaux, within savanna areas.

Grasslands are historically fire-prone and thus fire is a vital element in their maintenance (Bainbridge 1993; O’Connor and Bredenkamp 1997). Fire is known to have many effects on the vegetation and can lead to changes in species composition (Belsky 1992; O’Connor 1994; Pfab and Witkowski 1999; Uys et al. 2004; Kraaij 2010; Ghebrehiwot et al. 2011), structure (Bond and Keeley 2005; Shackleton 2007; Staver et al. 2009) and productivity (Fynn et al. 2003; Buis et al. 2009).

The negative effects of wild fires have prompted the legal requirement of the annual burning of fire-breaks in South Africa. Fire-breaks can comprise up to 10% of the surface area of a property and are required to be burnt by July (O’Connor et al. 2004). The timing, frequency and location of fire-breaks are usually fixed, so that the same area is constantly subjected to the impact of fire at the same time of the year. Fire-breaks can create an ecological mosaic, as most other burning takes place in early spring. Although the effects of fire have often been studied in different habitats in South Africa, the specific ecological effects of fire-breaks have been mostly ignored (O’Connor et al. 2004).

The effects of fire on the plant diversity in southern African natural grasslands are poorly understood because the focus in the past has mostly been on the dominant grass species needed for livestock production (Uys et al. 2004). Although the grasses constitute the greatest biomass in grasslands, the diversity of grasslands is found primarily in the forb richness (Uys et al. 2004). Some studies have shown that plant diversity in southern African grasslands is not, or at most marginally, affected by the fire regime (Uys et al. 2004; O’Connor et al. 2010),
whereas others indicate that plant diversity is affected by the occurrence and frequency of fire (de V Booyesen and Tainton 1984; O’Connor and Everson 1998; Fynn et al. 2004; Joubert et al. 2014; Duigan 2015).

The aim of this study was to determine the effects of annual burning of fire-breaks on plant species diversity, composition, range condition and soil properties in the grassland association on the Loskop Dam Nature Reserve.

Materials and methods

Study site

The study was conducted in the Loskop Dam Nature Reserve (LDNR) (25°22′ to 25°31′ S, 29°10′ to 29°24′ E; Eksteen 2003), which is located in the Olifants River valley about 55 km north of Middelburg in Mpumalanga (Figure 1) and covers approximately 23 000 ha.

The reserve falls in the summer rainfall region with moderate to very hot summers and moderate winters. About 84% of the total rainfall occurs in the months October to March as short, high-intensity thundershowers with severe lightning and strong southwesterly winds (Theron 1973; Eksteen 2003; Mucina and Rutherford 2006). The mean annual rainfall for the reserve is 718 mm (Theron 1973).

The reserve is characterised by broken terrain, with deep ravines and steep slopes with few plains and plateaux. Altitude ranges from 991 to 1 420 m above sea level. The major rivers in the reserve are the Olifants River and the Krantzspur, which both flow into the dam. Geologically, the reserve is underlain by the Waterberg Group and Rooiberg Group of the Bushveld Complex, Transvaal Supergroup (Eriksson and Vos 1979). The study sites were located in the Fa7 and Ib10 Land types. The Fa7 Land Type comprises very shallow Glenrosa and Mispah soil forms, whereas the Ib10 Land Type is dominated by the Mispah soil form (Land Type Survey Staff 1987).

The area occurs on the transition between the Grassland and Savanna Biomes. In the valleys the vegetation is typical of the Savanna Biome, whereas the plateau vegetation is typical of the Grassland Biome and resembles Bankenveld vegetation (Eksteen 2003). Mucina and Rutherford (2006) identified three savanna vegetation types on the reserve: Loskop Mountain Bushveld, Loskop Thornveld and Central Sandy Bushveld. However, the vegetation of the study sites bears more resemblance to the Rand Highveld Grassland of Mucina and Rutherford (2006), which is described as a highly variable landscape with extensive sloping plains. The species-rich vegetation type can be described as wiry, sour grassland with a high diversity of forbs. Rocky hills and ridges carry sparse savannoid woodlands with Protea caffra, Senegalia caffra (Acacia caffra) and Celtis africana being the most prominent.

Bachinger (2010) classified the vegetation on the higher-lying plateau region in the Nooitgedacht/DoornNek section of the Loskop Dam Nature Reserve into four communities and these communities were the focus of the current study (Table 1). Community 1, the Eragrostis nindensis–Tristachya leucothrix Grassland, was a typical Highveld grassland (hereafter referred to as Highveld grassland) whereas community 2, the Hyparrhenia hirta–Sporobolus africanus Grassland, represented a Highveld grassland on abandoned cropland that had last been cultivated in the 1980s (hereafter referred to as grassland on abandoned cropland). Community 3, Protea caffra–Tristachya leucothrix Grassland, and Community 4, Faurea saligna–Tristachya leucothrix Grassland (hereafter referred to as Protea caffra and Faurea saligna grassland, respectively), represented grassland communities with some savanna elements intermixed.

The Loskop Dam Nature Reserve is home to 70 mammal species. From a grazing point of view the most important herbivore species are: African buffalo (Syncerus caffer), blue wildebeest (Connochaetes taurinus), eland (Taurotragus oryx), giraffe (Giraffa camelopardalis), hippopotamus (Hippopotamus amphibius), impala (Aepyceros melampus), kudu (Tragelaphus strepsiceros), mountain reedbuck (Redunca fulvorufula), nyala (Tragelaphus angasii), plains zebra (Equus burchelli), sable antelope (Hippotragus niger), tsessebe (Damaliscus lunatus), waterbuck (Kobus ellipsiprymnus) and white rhinoceros (Ceratotherium simum).

Sampling techniques

An adaptation of a Whittaker plot (Shmida 1984) was used to sample plant composition and diversity in February and March 2009 and 2010. For each community, there were six paired plots, consisting of the fire-break and adjoining grassland plot. The fire-break plots were located approximately 10 m from the fence of the reserve along which the fire-breaks are burnt. The unburnt plots were located roughly 100 m from the fire-break plots, to ensure that they were not within the affected fire-break zone. For the purpose of this study the term ‘unburnt plots’ refers to the areas that are subjected to the normal reserve fire regime with an interval of approximately 3–4 years between fires. Each fire-break and unburnt pair within a community was approximately 100 m from the next pair. Each fire-break plot and corresponding unburnt plot experienced similar environmental conditions such as slope, topography and soil type. In community 1, fire-breaks were not burnt in the winter before sampling because a patch mosaic burn had occurred across the fire-breaks in the previous season. Each plot was only 10 × 10 m in size because of the limited space within the fire-break zone. The subsample plot layout consisted of a row of ten adjacent 1 m² plots, two adjacent 1 × 5 m (5 m²) plots, all nested inside two 5 × 10 m (50 m²) plots that together formed the 10 × 10 m (100 m²) quadrat.
Table 1: Biophysical features of the four communities investigated in the Loskop Dam Nature Reserve

| Community number | 1 |
|------------------|--|
| Full community name | **Eragrostis nindensis**–**Tristachya leucothrix** Grassland |
| Physiognomic description | Highveld grassland |
| Vegetation type (Mucina and Rutherford 2006) | Central Sandy Bushveld |
| Altitude (mean across six sites) | 1 355–1 456 m a.s.l. |
| Soil pH (mean across six sites) | 6.1 |
| Geology | Rhyolite |
| Land Type | Fa7 |
| Land-use history | Conservation |
| Biotic factors | Fire (both patch mosaic burning and fire-breaks) |
| Time since last fire in unbeurnt plots | 15–17 months |

| Community number | 2 |
|------------------|--|
| Full community name | **Hyparrhenia hirta**–**Sporobolus africanus** Grassland |
| Physiognomic description | Grassland on abandoned cropland |
| Vegetation type | Central Sandy Bushveld |
| Altitude (mean across six sites) | 1 355–1 456 m a.s.l. |
| Soil pH (mean across six sites) | 4.6 |
| Geology | Rhyolite |
| Land Type | Fa7 |
| Land-use history | Previously cultivated land |
| Biotic factors | Fire (both patch mosaic burning and fire-breaks) |
| Time since last fire in unbeurnt plots | >24 months |

| Community number | 3 |
|------------------|--|
| Full community name | **Protea caffra**–**Tristachya leucothrix** Grassland |
| Physiognomic description | Grassland (grassland with some savanna elements) |
| Vegetation type | Mix between Central Sandy Bushveld and Loskop Mountain Bushveld |
| Altitude (mean across six sites) | 1 355–1 456 m a.s.l. |
| Soil pH (mean across six sites) | 4.9 |
| Geology | Rhyolite |
| Land Type | Fa7 |
| Land-use history | Conservation |
| Biotic factors | Harvesting of thatching grass |
| Time since last fire in unbeurnt plots | >24 months |

| Community number | 4 |
|------------------|--|
| Full community name | **Faurea saligna**–**Tristachya leucothrix** Grassland |
| Physiognomic description | Grassland (grassland with some savanna elements) |
| Vegetation type | Mix between Central Sandy Bushveld and Loskop Mountain Bushveld |
| Altitude (mean across six sites) | 1 355–1 456 m a.s.l. |
| Soil pH (mean across six sites) | 5.2 |
| Geology | Rhyolite |
| Land Type | Ib10 |
| Land-use history | Conservation |
| Biotic factors | Fire (both patch mosaic burning and fire-breaks) |
| Time since last fire in unbeurnt plots | >24 months |

The presence of each species encountered in a quadrat was noted and a percentage cover value given for each species in the 100 m² quadrat. Because each column contained a list of all species present in that quadrat it was possible to calculate the number of species present in a quadrat of a different size than actually measured, e.g. 10 m².

One soil sample was taken randomly within each plot to a depth of 10 mm. The soil was analysed for pH (deionised water), soil resistance, carbon (Walkley–Black method), nitrogen (acid digestion) and phosphorous (acid digestion) content by the Soil Laboratory, Department of Plant Production and Soil Science, University of Pretoria.

**Data analysis**

The following parameters were calculated and analyses performed:

- Species richness (S), species evenness (E), the Shannon–Wiener index of diversity (H') and the Simpson index of diversity (D) were calculated for each 100 m² fire-break and unbeurnt plot using PC-ORD 5.0 (MJM Software Design, Gleneden Beach, OR, USA). In addition, the total number of species (species richness) within ten 1 m², two 5 m², and two 50 m² quadrats was calculated for the fire-break and unbeurnt plots in a community.

- For each plot the total number of species for five plot sizes (1 m², 5 m², 10 m², 50 m² and 100 m²) was calculated. These values were used to construct species–area curves (SAC), using the power function (Connor and McCoy 1979; Tjerve 2003) for all fire-break and unbeurnt plots in a community. Statistically significant differences between the slope and intercept values of the curves were analysed by Analysis of Covariance (Quinn and Keough 2002) with Graphpad Prism 4.0 (San Diego, CA, USA; http://www.graphpad.com).

- Species accumulation curves (SACc) were constructed in EstimateS 9 (Colwell 2013) using 100 randomisations of the floristic data for the 100 m² fire-break and unbeurnt plots in a community.

- The Jaccard index of similarity (Mueller-Dombois and Ellenberg 1974) was used to determine the floristic similarity between the species in the fire-break and unbeurnt plots in a community. For this analysis the species of all six fire-break plots in a community were combined and compared to the pooled species of the six unbeurnt plots.

- In addition, a similarity percentage analysis (SIMPER; Clarke 1993) was performed in PAST 3.02 (Hammer 2014) to identify species contributing most to the differences between fire-break and unbeurnt plots.

- The grazing value of the herbaceous layer within each community was determined using the range condition index as set out by Bothma et al. (2004).

- Ordinations of the floristic data were performed using correspondence analysis (CA) in Canoco 4.52 (Wageningen, The Netherlands) to determine which species were associated with fire-breaks and which with unbeurnt vegetation. Based on the gradient lengths, as advised by Lepš and Šmilauer (2003) either a linear model or unimodal model would have been appropriate. A canonical correspondence analysis (CCA) was used to determine species, plot and environmental interactions.

- After testing for normality of the data distribution (Kolmogorov–Smirnov test), all paired t-tests for significant differences were conducted in STATISTICA 9.0 (StatSoft, Tulsa, OK, USA).
Results

Diversity
The fire-break plots had significantly higher mean values than the unburnt plots for the Shannon–Wiener index ($p = 0.01$), Simpson index ($p = 0.01$) and evenness ($p = 0.02$) only for the grassland on abandoned cropland (community 2) (Figure 2). At the different plot sizes, the fire-break plots had a higher species richness than the unburnt plots, in the case of the grassland on abandoned cropland at two sizes (1 m$^2$: $p < 0.01$; 50 m$^2$: $p = 0.04$) and for the Highveld grassland at the 5 m$^2$ plot size ($p < 0.01$) (Figure 3).

In none of the communities did the slope of the fire-break and unburnt species–area curves (SAC) differ significantly (community 1: $p = 0.91$; community 2: $p = 0.91$; community 3: $p = 0.97$; community 4: $p = 0.24$). However, the intercept of the SAC was significantly higher for the fire-break curve than the unburnt curve in the Highveld grassland (community 1) and grassland on abandoned cropland (community 2) ($p < 0.01$ in both instances) (Figure 4).

The SACs showed that with the exception of community 1, fire-breaks accumulated more species across the six plots surveyed per community than the unburnt areas (Figure 5).

Species composition
Three of the four communities exhibited a clear separation between the fire-break and unburnt plots in the CA ordination (Figure 6, Supplementary Appendix S1). In community 1, which did not show a distinction between the fire-break and unburnt plots, fire-breaks were not burnt in the year before sampling.

In the other three communities, specific fire-break and unburnt species assemblages could be identified. In the grassland on abandoned cropland *Melinis nerviglumis*, *Themeda triandra*, *Eragrostis racemosa*, *Heteropogon contortus* and *Schizachyrium sanguineum* were associated with fire-break plots and *Eragrostis curvula*, *Eragrostis chioromelas* and *Hyparrhenia hirta* with the unburnt plots (Supplementary Appendix S1). In the Protea caffra grassland *Themeda triandra*, *Diheteropogon amplectens*, *Eragrostis curvula* and *Andropogon schirensis* had affinities with the fire-break plots, whereas *Tristachya leucothrix*, *Setaria sphacelata*, *Trachypogon spicatus* and *Bewsia biforna* had affinities with the unburnt plots (Supplementary Appendix S1).

The SIMPER analysis (Table 2) revealed that the species making the largest contribution to the dissimilarity were *Themeda triandra*, *Tristachya leucothrix*, *Hyparrhenia hirta* and *Setaria sphacelata*, with *Schizachyrium sanguineum* being prominent in the grassland on abandoned cropland.

Most of the species richness was contained in the forb species (mean 60.7% of all species at a site) (Table 3). Grasses contributed to approximately one-quarter of all species at a site (mean 25.2%), with the contribution of woody species and geophytes both less than 10%. The Jaccard index of similarity between the fire-break and unburnt plots revealed that the grass species were significantly more similar (63.1%) than the other growth forms.

Figure 2: Mean values for (a) species richness, (b) Shannon–Wiener index, (c) Simpson’s index and (d) species evenness for fire-break and unburnt 100 m$^2$ plots in four grassland communities on the Loskop Dam Nature Reserve. An asterisk denotes a statistically significant difference between the fire-break and unburnt value. Sites: 1 = Community 1, Highveld grassland; 2 = Community 2, grassland on abandoned cropland; 3 = Community 3, Protea caffra grassland; and 4 = Community 4, Faurea saligna grassland.
Figure 3: Mean species richness in the 1 m², 5 m², 50 m² and 100 m² quadrats in the (a) Highveld grassland (community 1), (b) grassland on abandoned cropland (community 2), (c) *Protea caffra* grassland (community 3) and (d) *Faurea saligna* grassland (community 4). An asterisk denotes a statistically significant difference between the fire-break and unburnt value.

Figure 4: Species–area curve applying the power function to the diversity data of the fire-break and unburnt plots for (a) Highveld grassland (community 1), (b) grassland on abandoned cropland (community 2), (c) *Protea caffra* grassland (community 3) and (d) *Faurea saligna* grassland (community 4).
Bachinger, Brown and van Rooyen

Forb (37.8%), geophyte (35.56%) and woody species similarity (29.2%) were far lower than that of the grasses, but did not differ significantly from each other. Alien invasive species did not make a large contribution towards species richness and no significant differences were found between the mean number of alien species per 100 m² in fire-break and unburnt plots in any of the communities (community 1: \( p = 0.2354 \); community 2: \( p = 0.6109 \); community 3: \( p = 0.2031 \); community 4: \( p = 0.2956 \)).

The range condition index was significantly higher in the fire-break than the unburnt plots only in the *Protea caffra* grassland \((p < 0.01; \text{Table 4})\).

**Discussion**

**Diversity**

With the exception of the grassland on abandoned cropland, fire-breaks did not have a marked effect on the plant diversity parameters investigated in the current study. In this community species evenness, Shannon–Wiener index, Simpson’s index and species richness (1 and 5 m² plot) were significantly higher in the fire-break than the unburnt treatments.

The results of the current study support O’Connor et al. (2004) who also found that fire-breaks had only a slight effect on plant diversity in the montane grasslands of the southern Drakensberg. The differences reported by them were a slightly greater species evenness, graminoid density and a tendency for higher total species richness on fire-breaks when compared to the unburnt vegetation. Similarly, Uys et al. (2004) contended that plant diversity in southern African grasslands seems to be independent of fire regime. However, the positive impact on plant diversity in the fire-break area of the grassland on abandoned cropland indicates that fire does affect plant diversity in grassland undergoing succession.

The lack of any significant difference in slope values for the species–area curves for the fire-break and unburnt plots indicated that the species were accumulating at a similar rate from the smallest quadrat to the largest quadrat whether the site had been burnt or not. However, the
significantly higher intercepts in the fire-breaks than the unburnt plots showed that the fire-break plots had a larger number of species in the smaller plot sizes in two communities. At the community level, the species accumulation curves displayed a higher species richness in fire-breaks than unburnt vegetation, except for community 1 where the fire-break had not been burnt in the previous winter.

Species composition
In spite of the distinct species assemblages for fire-breaks and unburnt plots, it was difficult to single out specific species that were consistently associated with either fire-breaks or unburnt plots. Amongst the grasses *Themeda triandra* was the only species that was consistently associated with fire-breaks and *Schizachyrium sanguineum* was associated with fire-breaks in two communities. *Tristachya leucothrix*, *Setaria sphacelata*, *Bewsia biflora* and *Brachiaria serrata* were generally associated with the unburnt vegetation. These species are typical climax grass species associated with high-altitude grasslands (Bredenkamp and Brown 2003). Amongst the forbs *Zornia linearis* appeared to be more common in the fire-breaks and *Kohautia* sp. in the unburnt plots.

O’Connor et al. (2004) reported that fire-breaks had a notable influence on species composition in the montane grasslands of the southern Drakensberg. They found that *Themeda triandra* showed no affiliation to fire, which counters the findings of the current study. Several other studies have, however, shown that *Themeda triandra* increases with a short fire interval (Bond and van Wilgen 1996; O’Connor and Everson 1998; Novellie and Kraaij 2010). O’Connor et al. (2004) reported that *Eragrostis racemosa* and *Brachiaria serrata* were more abundant on fire-breaks. However, in the current study, *Eragrostis*
Bacinger, Brown and van Rooyen

 showed no consistent fire preference and was associated with the unburnt plots. The abundance of in the unburnt plots in the current study agrees with reports that this species prefers longer intervals between burns (de V Booysen and Tainton 1984; Bond and van Wilgen 1996; O'Connor and Everson 1998; van Oudtshoorn 2009).

The difference in species composition between fire-breaks and unburnt plots was evident when examining the similarity indices, which indicated that only approximately 44% of all species were common to fire-break and unburnt vegetation (Table 3). The difference in species composition was primarily contained in the forb species, which made up the bulk of the species and had a low similarity percentage. Grass species showed a significantly higher similarity than the forb, geophyte and woody growth forms, supporting the findings of O'Connor et al. (2004) in the montane grasslands of the southern Drakensberg.

Although alien species are often favoured by disturbance, fire-breaks did not increase the occurrence of alien species. The changes in species composition brought about by the annual burning were also not having a negative impact on the grazing value of the grassland as reflected by the range condition index. Thus, although the grazing value of the fire-breaks remained unchanged or even improved, the unburnt grassland seemed to have a higher proportion of so-called climax grass species than the fire-breaks (Bredenkamp and Brown 2003).

In community 1, where the fire-breaks were not burnt in the winter before sampling, the results differed from the other communities in several respects. Firstly, the ordinations did not show a clear separation between the fire-break and unburnt plots and, consequently, no clear distinction could be made between species associated with and without fire. Secondly, the similarity in species composition between fire-break and unburnt vegetation was the highest in this community with 52.1% of all species being common to fire-break and unburnt plots. Thirdly, the species accumulation curves of unburnt plots were higher than the fire-break.

### Table 2: SIMPER analysis indicating the species and their percentage contribution towards the first 50% of the dissimilarity between fire-break and unburnt plots

| Species                  | Contribution (%) | Ranking |
|--------------------------|------------------|---------|
| Bewsia biflora           | 5.74             | 5       |
| Eragrostis chloromelas    | 4.92             | 6       |
| Heteropogon contortus     | 8.15             | 3       |
| Hyparrhenia hirta         | 19.40            | 1       |
| Loudelia simplex          | 12.50            | 2       |
| Protea caffra             | 13.58            | 2       |
| Schizachyrium sanguineum  | 12.41            | 3       |
| Themeda triandra          | 10.75            | 2       |
| Trachypogon spicatus      | 8.00             | 4       |
| Tristachya leucothrix     | 13.00            | 1       |
| Total                    | 50.56            | 51.81   |

### Table 3: Floristic similarity between the species in the fire-break and unburnt plots in four communities on the Loskop Dam Nature Reserve. The value in brackets refers to the contribution (%) of the growth form to the total number of species. Values followed by the same superscript letter are not significantly different at $\alpha = 0.05$

| Community                 | Jaccard index (%) (contribution by growth form) |
|---------------------------|-----------------------------------------------|
|                           | Forb species | Grass species | Geophyte species | Woody species | All species |
| Community 1: Highveld grassland | 49.5 (62.3) | 67.6 (25.3) | 44.4 (6.2) | 22.2 (6.2) | 52.1 |
| Community 2: Grassland on abandoned cropland | 31.2 (59.1) | 66.7 (31.4) | 25.0 (3.8) | 33.3 (5.7) | 42.3 |
| Community 3: Protea caffra grassland | 32.6 (62.8) | 58.1 (22.6) | 28.6 (5.1) | 15.4 (9.5) | 36.5 |
| Community 4: Faurea saligna grassland | 37.9 (58.6) | 60.0 (21.6) | 44.4 (5.6) | 45.8 (14.2) | 44.4 |
| Mean (n = 4)               | 37.8 (60.7) | 63.1 (25.2) | 35.6 (5.2) | 29.2 (8.9) | 43.8 |

### Table 4: Grazing value (as reflected by the Range Condition Index) of the herbaceous layer in the fire-break and unburnt plots in four communities on the Loskop Dam Nature Reserve (n = 6)

| Community                 | Range Condition Index ± SE                  |
|---------------------------|---------------------------------------------|
|                           | Fire-break | Unburnt |
| Community 1: Highveld grassland | 64.7 ± 2.7 | 57.4 ± 2.3 |
| Community 2: Grassland on abandoned cropland | 58.4 ± 3.6 | 59.1 ± 2.9 |
| Community 3: Protea caffra grassland | 65.8 ± 4.2 | 41.0 ± 2.3 |
| Community 4: Faurea saligna grassland | 56.7 ± 5.6 | 60.7 ± 2.4 |
plots. These statements offer additional support that fire-breaks do not have a marked effect on plant diversity and that the differences in species composition depend on the continued annual burning of the fire-break zones.

Relationships between species and environmental variables
An increase in the carbon and nitrogen concentrations outside the fire-break zone, found in this study, supports the findings by O’Connor et al. (2004). Jones et al. (1990) also reported nitrogen increases with a decrease in fire frequency. However, Shackleton and Scholes (2000) found no differences in the soil nitrogen in terms of fire frequency and Trappell (1959) found no difference in soil carbon in annually burnt and unburnt areas. Fynn et al. (2003) reported that decreases in organic carbon were observed only in the surface soil layer (0–2 cm) and only under annual and biennial winter burning and biennial and triennial autumn burning. In contrast, total nitrogen content was decreased substantially to a depth of 6 cm by all burning treatments (Fynn et al. 2003).

In the current study pH was not associated with fire treatment. In contrast, O’Connor et al. (2004) found a slight decrease in pH on fire-breaks. This decline in pH has been suggested to be due to the increase in soil temperature in the fire-break zone as a result of the blackened soil surface, which increases absorption of solar radiation. This process leads to an increase in soil microbial activity, which in turn leads to mineralisation or oxidation of the organic constituents of the soil (Knapp et al. 1998; O’Connor et al. 2004).

Some of the reported negative effects of fire on soil include the increased potential for soil crusting, which leads to soil compaction and therefore disturbs the hydrological cycle (Mills and Fey 2004; Salvadogo et al. 2007). Annually burnt fire-breaks are, however, not as prone to soil erosion as biennially burnt grasslands, due to the fact that they are burnt in winter and therefore the canopy cover has a chance to recover before the spring rains.

Succession on grassland on abandoned cropland
The positive effect of fire-breaks on plant diversity on the abandoned cropland, which was last cultivated in the 1980s (community 2), was notable. There is not a vast amount of research on the restoration of formerly cultivated lands, especially in the Highveld grassland (van Rheede van Oudtshoorn 2007) and much of the knowledge is still based on the studies conducted at Frankenwald (Roux 1969). The successful rehabilitation of old fields is becoming of increasing importance to the conservation of grassland (van Rheede van Oudtshoorn 2007) because worldwide there is an increasing trend of land abandonment as a result of environmental and socio-economic changes (Cramer and Hobbs 2007; Cramer et al. 2007).

Although some plant communities have the ability to recover from a disturbance through the process of natural succession, many components of community structure

Table 5: Soil properties for all the fire-break and unburnt plots of four communities in the grassland association on the Loskop Dam Nature Reserve

| Community                              | Fire-break/Unburnt | Carbon (%) | Phosphorus (%) | Nitrogen (%) | pH   | Resistance (Ohm) |
|----------------------------------------|--------------------|------------|----------------|--------------|------|------------------|
| Community 1: Highveld grassland        | Fire-break         | 1.77       | 0.03           | 0.09         | 4.69 | 2 367            |
|                                        | Unburnt            | 2.11       | 0.03           | 0.13         | 6.08 | 1 767            |
| Community 2: Grassland on abandoned cropland | Fire-break         | 1.61       | 0.03           | 0.07         | 4.57 | 3 150            |
|                                        | Unburnt            | 2.68       | 0.03           | 0.12         | 4.62 | 2 583            |
| Community 3: Protea caffra grassland   | Fire-break         | 1.79       | 0.03           | 0.09         | 4.93 | 2 858            |
|                                        | Unburnt            | 2.17       | 0.03           | 0.09         | 4.89 | 2 225            |
| Community 4: Faurea saligna grassland  | Fire-break         | 1.37       | 0.03           | 0.06         | 5.14 | 3 533            |
|                                        | Unburnt            | 1.82       | 0.02           | 0.08         | 5.23 | 2 633            |
| All communities                        | Fire-break         | 1.63       | 0.03           | 0.08         | 4.83 | 2 977            |
|                                        | Unburnt            | 2.20*      | 0.03           | 0.10         | 4.95 | 2 302*           |

* Values significantly different at $\alpha = 0.05$
are slow to return without the aid of human involvement (Pywell et al. 2002). This applies to old cultivated lands where the chemical and physical structure of the soil has been severely altered and the natural seed bank has been removed (Aronson et al. 1993). Abandoned croplands in the grasslands of Mpumalanga, are often dominated by *Hyparrhenia* species. These *Hyparrhenia*-dominated grasslands are found to be stable for a very long time (Roux and Warren 1963; van Rheede van Oudtshoorn 2007) and the progression from a *Hyparrhenia hirta*-dominated stage to a more ecologically diverse stage is seldom encountered. These *Hyparrhenia hirta*-dominated grasslands usually show low species richness, with few other species being able to establish and survive in the shade created by this tall grass (van Oudtshoorn et al. 2011; van Rheede van Oudtshoorn 2007). This highly competitive species is one of the first species to colonise old lands as its seed is dispersed by wind. The most important grass species associated with *Hyparrhenia hirta*-dominated grassland are either pioneers or early successional species such as *Aristida congesta*, *Cynodon dactylon*, *Eragrostis curvula* and *Eragrostis racemosa* (van Rheede van Oudtshoorn 2007). Forbs are not abundant, but those that do occur include *Anthospermum rigidum*, *Crabbea angustifolia*, *Conyza podocephala* and *Helichrysum rugulosum* (van Rheede van Oudtshoorn 2007).

In a study in the Suikerbosrand Nature Reserve, Gauteng, South Africa, to investigate reseeding methods for the restoration of abandoned cultivated land in the Highveld region, it was found that where vegetation had been removed by ripping of the soil more perennial plant species could establish (van Rheede van Oudtshoorn 2007). In the current study it was thought that one of the reasons behind the increased plant diversity in the fire-break zone of the grassland on abandoned cropland was due to the fact that fire decreased the strong growth of *Hyparrhenia hirta*, thereby making more light and space available to other plant species for seedling recruitment and establishment.

**Conclusion**

There is a dearth of knowledge on the effects of fire-breaks on species composition, diversity and soil properties. This study demonstrated that fire-breaks do not have a negative effect on plant diversity. The only community to display positive results, in terms of its plant diversity parameters and species richness at different spatial scales, was the grassland on abandoned cropland. At this site, fire-breaks were having a positive influence by reducing *Hyparrhenia hirta* dominance and increasing plant diversity.

Fire-breaks were changing the species composition of the grassland association on LDNR, as three of the four grassland sites illustrated a clear separation between the fire-breaks and unburnt plots. There was, however, not a strong correlation between specific species and the fire-break or unburnt plots. A few species stood out, among them *Themeda triandra* and *Tristachya leucothrix*, known for their association with and without fire, respectively. The change in species composition was not negatively affecting range condition and at one site the range condition was significantly higher in the fire-break than in the adjacent unburnt vegetation.

Fire-breaks were also altering the soil properties that were examined in this study. These changes were slightly negative, as carbon and nitrogen concentrations were lower in the fire-break zones. Although these concentrations were lower in the fire-break plots, the amounts by which they decreased were not sufficiently large to state with any certainty that fire-breaks were degrading the soil.

It is important to note that the results of the current study refer to fire-breaks and should not be extrapolated to block-burns. Fire-breaks are implemented mainly in the beginning to middle of the dry season (winter). The length of fire-breaks can be many kilometres long, traversing different plant communities along the perimeter of the land. Thus the trampling and grazing effects of animals congregating on these fire-breaks, as opposed to block burns, are mitigated by the animals being evenly dispersed along the burnt area. Because animals generally move to the larger block-burn areas in spring, fire-breaks have the remainder of the growing season to recover.

This paper provides insight into some aspects of the effects of annual fire-breaks on grassland vegetation. However, the effect on production, plant community dynamics as well as the faunal component should be studied further to gain a full understanding of the effects of annual burns on grasslands. Furthermore, a finer subdivision of the forb component into annual, biennial and perennial forbs needs to be considered in future studies. Similar studies should also be undertaken in savanna communities to assist land managers in developing scientifically supported burning strategies. The use of annual burning to stimulate the vegetation of abandoned cropland also needs to be further investigated and could have potential benefits for conservation areas in which old fields have been brought back into reserve boundaries.

From a management perspective, the annual burning of fire-breaks on LDNR appears to be a sustainable management tool as it is not adversely affecting the plant diversity or range condition in this grassland association.

**Acknowledgements** — The authors would like to express their appreciation towards the Mpumalanga Tourism and Parks Agency and the Loskop Dam Nature Reserve management for allowing us to make use of their facilities. The University of Pretoria, the Applied Behavioural Ecology and Ecosystem Research Unit at UNISA and the National Research Foundation are thanked for financial assistance.

**References**

Aronson J, Floret C, Ovalle C, Pontanier R. 1993. Restoration and rehabilitation of degraded ecosystems in arid and semi-arid lands: A view from the South. *Restoration Ecology* 1: 8–17.

Bachinger LM. 2010. Effects of fire-breaks on plant diversity in the Loskop Dam Nature Reserve, Mpumalanga, South Africa. MSc dissertation, University of Pretoria, South Africa.

Bainbridge WR. 1993. Management of mountain catchment grassland with special reference to the Natal Drakensberg. In: van der Sijde HA (ed.), *South African forestry handbook*. Pretoria: Southern African Institute of Forestry. pp 657–690.

Belsky AJ. 1992. Effects of grazing, competition, disturbance and fire on species composition and diversity in grassland
communities. *Journal of Vegetation Science* 3: 187–200.

Bond WJ, Keeley JE. 2005. Fire as a global herbivore: the ecology and evolution of flammable ecosystems. *Trends in Ecology and Evolution* 20: 367–374.

Bond WJ, van Wilgen BW. 1996. *Fire and plants*. London: Chapman and Hall.

Bothma J du P, van Rooyen N, van Rooyen MW. 2004. Using diet and plant resources to set wildlife stocking densities in African savannas. *Wildlife Society Bulletin* 32: 840–851.

Bredenkamp GJ, Brown LR. 2003. A reappraisal of Acocks’ Bankenveld: origin and diversity of vegetation types. *South African Journal of Botany* 69: 7–26.

Buis GM, Blair JM, Burklepelle DE, Burns CE, Chamberlain AJ, Chapman PL, Collins SL, Fynn RWS, Govender N, Kirkman KP, Smith MD, Knapp AK. 2007. Controls of aboveground net primary production in mesic savanna grasslands: an inter-hemispheric comparison. *Ecosystems* 12: 982–995.

Clarke KR. 1993. Non-parametric multivariate analyses of changes in community structure. *Austral Ecology* 18: 117–143.

Colwell RK. 2013. EstimateS: Statistical estimation of species richness and shared species from samples. Version 9. User’s guide and application available at: http://purl.oclc.org/estimates.

Connor EF, McCoy ED. 1979. The statistics and biology of the species-area relationship. *American Naturalist* 113: 791–833.

Cramer VA, Hobbs RJ (eds). 2007. *Species-Area Relationship*. New York: John Wiley and Sons.

Cramer VA, Hobbs RJ (eds). 2007. *EstimateS: Statistical estimation of species area relationship*. http://purl.oclc.org/estimates.

Cramer VA, Hobbs RJ. 2007. What’s new about old fields: dynamics and evolution of vegetation types. *South African Journal of Botany* 76: 550–557.

Cramer VA, Hobbs RJ (eds). 2007. *EstimateS: Statistical estimation of species area relationship*. Guide and application available at: http://purl.oclc.org/estimates.

Cramer VA, Hobbs RJ. 2007. What’s new about old fields? Land abandonment and ecosystem assembly. *Trends in Ecology and Evolution* 23: 104–112.

de V Booyse P, Tainton NM. 1984. *Loskop Dam Nature Reserve information booklet*. Departmental report. Middelburg: Mpumalanga Nature Conservation.

Eksteen J. 2003. *Loskop Dam Nature Reserve information booklet and management plan*. Departmental report. Middelburg: Mpumalanga Nature Conservation.

Eriksson KA, Vos RG. 1979. A fluvial fan depositional model for Middle Proterozoic red beds from the Waterberg Group, South Africa. *Precambrian Research* 9: 169–188.

Fairbanks DHK, Thompson MW, Vink DE, Newby TS, van den Berg HM, Everard DA. 2000. The South African land-cover characteristics database: a synopsis of the landscape. *South African Journal of Science* 96: 65–82.

Fynn RWS, Haynes RJ, O’Connor TG. 2003. Burning causes long-term changes in soil organic matter content of a South African grassland. *Soil Biology and Biochemistry* 35: 677–687.

Fynn RWS, Morris CD, Edwards TJ. 2004. Effect of burning and mowing on grass and forb diversity in a long-term grassland experiment. *Applied Vegetation Science* 7: 1–10.

Ghebrehiwot HM, Kulkarni MG, Light ME, Kirkman KP, van Staden J. 2011. Germination activity of smoke residues in soils following a fire. *South African Journal of Botany* 77: 716–724.

Hammer Ø. 2014. PAST: PAleontological Statistics Version 3.02. Reference manual. Oslo: Natural History Museum, University of Oslo.

Jones CL, Smithers NL, Scholes MC, Scholes RJ. 1990. The effect of fire frequency on the organic components of a basaltic soil in the Kruger National Park. *South African Journal of Plant and Soil* 7: 236–238.

Joubert L, Pryke JS, Samways MJ. 2014. Annual burning drives plant communities in remnant grassland ecological networks in an afforested landscape. *South African Journal of Botany* 92: 126–133.

Knapp AK, Briggs JM, Blair JM, Turner CL. 1998. Patterns and controls of above ground net primary production in tallgrass prairie. In: Knapp AK, Briggs JM, Hartnett DC, Collins SL (eds), *Grassland dynamics: long-term ecological research in tallgrass prairie*. Oxford: Oxford University Press. pp 193–221.

Kraaj T. 2010. Changing the fire management regime in the renosterveld and lowland fynbos of the Bontebok National Park. *South African Journal of Botany* 76: 550–557.

Land Type Survey Staff. 1987. *Land types of the map 2528 Pretoria*. Memoirs on the Agricultural Natural Resources of South Africa no. 8. Pretoria: ARC–Institute for Soil, Climate and Water.

Lepi J, Šmilauer P. 2003. *Multivariate analysis of ecological data using CANOCO*. Cambridge: Cambridge University Press.

Mills AJ, Fey MV. 2004. Frequent fires intensify soil crusting: physicochemical feedback in the pedoderm of long-term burn experiments in South Africa. *Geoderma* 121: 45–64.

Mucina L, Rutherford MC (eds). 2006. *The vegetation of South Africa, Lesotho and Swaziland*. Strelitzia 19. Pretoria: South African National Biodiversity Institute.

Mueller-Dombois D, Ellenberg H. 1974. *Aims and methods of vegetation ecology*. New York: John Wiley and Sons.

Novelle P, Kraaj T. 2010. Evaluation of *Themeda triandra* as an indicator for monitoring the effects of grazing and fire in the Bontebok National Park. *Koedoe* 52: Art. #977, 5 pages.

O’Connor TG. 1994. Composition and population response of an African savanna grassland to rainfall and grazing. *Journal of Applied Ecology* 31: 155–171.

O’Connor TG, Bredenkamp GJ. 1997. Grassland. In: Cowling RM, Richardson DM, Pierce SM (eds). *Vegetation of South Africa*. Cambridge: Cambridge University Press. pp 215–257.

O’Connor TG, Everson TM. 1998. Population dynamics of perennial grasses in African savanna and grassland. In: Cheplick GP (ed.), *Population biology of grasses*. Cambridge: Cambridge University Press. pp 333–355.

O’Connor TG, Uys RG, Mills AJ. 2004. Ecological effects of fire-breaks in the montane grasslands of the southern Drakensberg, South Africa. *African Journal of Range and Forage Science* 21: 1–9.

O’Connor TG, Kuyler P, Kirkman KP, Corcoran B. 2010. Which grazing management practices are most appropriate for maintaining biodiversity in South African grasslands? *African Journal of Range and Forage Science* 27: 67–76.

Pfab MF, Witkowski ETF. 1999. Fire survival of the Critically Endangered succulent, *Euphorbia clivicola* R.A. Dyer – fire avoider or fire tolerant. *African Journal of Ecology* 37: 249–257.

Pywell RF, Bullock JM, Hopkins A, Walker KJ, Sparks TH, Burkes MJW, Peet S. 2002. Restoration of species-rich grassland on arable land: assessing the limiting processes using a multi-site experiment. *Journal of Applied Ecology* 39: 294–309.

Quinn GP, Keough MJ. 2002. *Experimental design and data analysis for biologists*. Cambridge: Cambridge University Press.

Roux E (ed.). 1969. *Grass: a story of Frankenwald*. Cape Town: Oxford University Press.

Roux ER, Warren M. 1963. Plant succession on abandoned fields in central Oklahoma and in the Transvaal Highveld. *Ecology* 44: 576–579.

Salvadogo P, Sawadogo L, Tiveau D. 2007. Effects of grazing intensity and prescribed fire on soil physical and hydrological properties and pasture yield in the savanna woodlands of Burkino Faso. *Agriculture, Ecosystems and Environment* 118: 80–92.

Shackleton CM. 2007. The effects of fire on post-fire seed germination of selected savanna woody species. *African Journal of Ecology* 45: 545–549.

Shackleton CM, Scholes RJ. 2000. Impact of fire frequency on woody community structure and soil nutrients in the Kruger National Park. *Koedoe* 43: 75–81.

Shmida A. 1984. Whittaker’s plant diversity sampling method. *Israel Journal of Botany* 33: 41–46.

Staver AC, Bond WJ, Stock WD, van Rensburg SJ, Waldram MS.
2009. Browsing and fire interact to suppress tree density in an African savanna. *Ecological Applications* 19: 1909–1919.
Theron GK. 1973. ‘n Ekologiese studie van die plantegroei van die Loskopdam-natuurreserveat. DSc thesis, University of Pretoria, South Africa.
Tjørve E. 2003. Shapes and functions of species-area curves: a review of possible models. *Journal of Biogeography* 30: 827–835.
Trapnell CG. 1959. Ecological results of woodland and burning experiments in northern Rhodesia. *Journal of Ecology* 47: 129–168.
Uys RG, Bond WJ, Everson TM. 2004. The effect of different fire regimes on plant diversity in southern African grasslands.
*Biological Conservation* 118: 489–499.
van Oudtshoorn FP. 2009. *Guide to grasses of southern Africa*. Pretoria: Briza Publications.
van Oudtshoorn FP, Brown LR, Kellner K. 2011. The effect of re-seeding methods on secondary succession during cropland restoration in the Highveld region of South Africa. *African Journal of Range and Forage Science* 28: 1–8.
van Rheede van Oudtshoorn FP. 2007. The evaluation of various reseeding methods for restoring old croplands in the highveld region of South Africa. MSc dissertation, Johannesburg: University of South Africa.