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RESEARCH



Alien tree invasion into a South African montane grassland ecosystem: impact of *Acacia* species on rangeland condition and livestock carrying capacity

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ABSTRACT

The degree to which invasive *Acacia* species affect South Africa's livestock production has received little attention. We investigated the ecological impacts of *Acacia mearnsii* invasion on forage quality and quantity and on soil resources, along *A. mearnsii* invasion gradients, on South African rangelands and the subsequent conditions following clearing. Grazing capacity was reduced by 72% in densely invaded sites, whereas clearing improved grazing capacity by 66% relative to densely invaded sites within 5 years. In densely invaded sites total grass species basal cover was reduced by up to 42%. As a result, *A. mearnsii* reduced grazing capacity, from 2 to 8 ha required to support one large stock unit (ha/LSU) in uninvaded and densely invaded sites, respectively. Soil moisture content was lower in densely invaded sites compared with lightly invaded and cleared sites. Plant litter increased from 1.3% to 4.2%, carbon content of the soil increased from 2.0% to 4.0% and nitrogen concentrations increased from 0.1% to 0.2% in response to invasion by *A. mearnsii*. Clearing resulted in improved grazing capacity within 5 years. These results also showed that, if left uncontrolled, wattle species can reduce livestock carrying capacity within montane grasslands in South Africa.

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1. Introduction

Invasion of rangelands by alien plants is a threat to livestock production in South Africa (Wise et al. 2012; Shackleton et al. 2015; Ndhlovu et al. 2016; Stafford et al. 2016). Livestock production in South Africa is predominantly rangeland-based, hence rangeland condition is crucial for sustainable forage production (Cousins 1999). Fast-growing invasive *Acacia* species use more water than grasslands and indigenous tree species and alter nutrient flows, potentially reducing the ability of rangeland ecosystems to provide forage (De Neergaard et al. 2005; Moyo and Fatunbi 2010). Australian *Acacia* species have the potential to convert grassland into dense woodlands through the displacement of grasses and shrubs (De Neergaard et al. 2005). However, little work has been done to quantify the impacts of invasive alien plant invasions on livestock production in South Africa, particularly at local scales (Ndhlovu et al. 2016; Stafford et al. 2016). This is important because it has clear implications for both land management and human wellbeing in rural areas.

Over the past years, work on understanding the impacts of invasive alien species on South African ecosystems and human wellbeing has been growing and still continues to grow (Richardson and Van Wilgen

2004; van Wilgen et al. 2008; Le Maitre et al. 2011). The Working for Water (WfW) programme is a national alien plant control programme run by the Department of Environment Affairs focussed on controlling the spread of invasive alien species (van Wilgen et al. 1998). The primary goal of the programme and the rationale behind its establishment was to secure threatened water resources through the clearing of invasive alien plants countrywide. At the same time the programme also addresses poverty through the creation of employment (van Wilgen et al. 1998; Jones et al. 2011). The project rests on a further premise that clearing of invasive alien plants will restore ecosystems to their initial conditions (van Wilgen et al. 2012).

Acacia mearnsii De Wilde (an introduced species of *Fabaceae* family from Australia) is one of the most prolific and extensive invaders of rangelands in the more mesic areas of South Africa, particularly in the Eastern Cape, KwaZulu-Natal and Mpumalanga Provinces (Nel et al. 2004; De Neergaard et al. 2005). Several studies have reported that the removal of *Acacia* species can facilitate re-colonisation by indigenous species and improve basal vegetation cover (Blanchard and Holmes 2008; Ruwanza et al. 2013). However, while some areas can recover following the removal of invasive alien plants without

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further interventions, in other areas active restoration is required to enhance ecosystem recovery (Beater et al. 2008; Gaertner et al. 2011; Le Maitre et al. 2011). In some cases, elevated levels of soil nutrients (Yelenik et al. 2007; Gaertner et al. 2011; Witkowski 2012) derived from nutrient rich litter, and N fixation in the case of legumes, can lead to the undesirable situation of reinvasion by the same and or other species after clearing (Beater et al. 2008; Witkowski and Garner 2008). Studies of the impacts of invasive alien plants and the recovery of rangelands after clearing can help quantify the damage caused by and benefits of clearing invasive alien plants (Ndhlovu et al. 2016). While there are several studies reporting on the benefits of clearing and the recovery of some ecosystem components (Yelenik et al. 2004, 2007; Blanchard and Holmes 2008; Gaertner et al. 2011; Witkowski 2012) these have not been conducted in the context of grazing capacities.

This study quantifies the impact of invasions by *A. mearnsii* and benefits associated with clearing operations on range condition and livestock production in the Amathole montane grassland of Stutterheim, South Africa. The objectives of the study were (i) to determine the effects of *A. mearnsii* density on indigenous grass species cover, and how this translates into impacts in forage quality and quantity; (ii) to determine the effects of *A. mearnsii* invasion on soil resources and conditions required to support grazing capacity and (iii) to determine the effects of clearing operations on grazing capacity.

2. Materials and methods

2.1. Study site

The study was conducted in the Stutterheim district of the Eastern Cape Province (32° 34' 0.84"S, 27° 25' 0.12"E), South Africa. Stutterheim has a 40-year mean annual rainfall of 752 mm, with most of the rain falling between October and March (O'reagain and Grau 1995). Average minimum and maximum temperatures range from 14 to 25°C, in summer, and 7 to 18°C, in winter. Mean annual rainfall of the period of interest (2005–2010 when clearing of invasions was undertaken) has been generally higher than the long term average (calculated for 40 years). The dominant vegetation type found here is Amathole Montane Grassland (Mucina et al. 2006) characterised as short grassland with a high species richness of forbs (especially *Helichrysum* and *Senecio* species). The dominant grass species include *Themeda triandra* (Forssk.), *Sporobolus africanus* (Poir.), *Eragrostis chloromelus* (Schrad.), *E. curvula* (Schrad.), *Herteropogon contortus* (L) and *Tristachya leucothrix* (Nees), while the karroid shrubs *Chrysocoma ciliata* (L) and *Felicia filifolia* (Vent.) are also prevalent. The most prominent

invaders into this vegetation type include *A. mearnsii* (De Wild.) and *Acacia dealbata* (Link)(Mucina et al. 2006). Geologically, the soils are derived from sedimentary rocks of the Beaufort Group and are characterised by weakly developed lithosols, solonchic soils interspersed with red clay (Mucina et al. 2006). Land use within this region is dominated by livestock production with numerous cattle and sheep stud farms, commercial livestock farms as well as communally based livestock (Kopke 1961). Stocking rates range between 2 and 40 ha/LSU under optimal and poor conditions, respectively (Trollope 1986). The area also supports forestry industries that include extensive *Pinus*, *Eucalyptus* and non-native *Acacia* species plantations. More than 10% of the landscape has been transformed to plantations and cultivation (Kopke 1961).

We investigated two levels of wattle tree invasion, lightly invaded and densely invaded, and compared these with uninvaded and cleared sites (Figure 1). For each of these four treatments five sites were examined ($n = 20$ sites) on the Stanhope, Siyothula and Krause's farms (Figure 1) within the Amathole montane grassland, (Mucina et al. 2006). Study sites were at least 200 m apart from each other. The initial clearing of *A. mearnsii* was done 5 years prior to assessment (cleared in 2006). Discussions with landowner confirmed the state of tree density prior to clearing as equivalent to that of densely invaded sites. Clearing operations involved the complete mechanical removal of *A. mearnsii* by cutting and the application of herbicide on cut stumps to prevent resprouting. Follow-up treatments were applied every year after the initial clearing, and sites were burnt after clearing to stimulate germination from the soil stored seed bank. We acknowledge that we do not have true replicates of cleared sites, as there were no other cleared sites in reasonable proximity, but it does not negate undertaking such work, but it does demand some caution in interpretation of the results. Field surveys were undertaken during the period December 2011 to February 2012.

2.2. Sampling and data collection

We investigated two levels of wattle tree invasion, lightly invaded and densely invaded, and compared these with uninvaded and cleared sites (Figure 1). Invaded sites (lightly and densely invaded sites) were visually assessed from Google Earth to estimate the aerial cover of wattle trees. For each of these four treatments, five sites were examined ($n = 20$ sites) on the Stanhope, Siyothula and Krause's farms (Figure 1) within the Amathole montane grassland (Mucina et al. 2006). Study sites were at least 200 m apart from each other. Uninvaded sites were selected to represent the current rangeland condition of the study area. To assess the degradation gradient of the

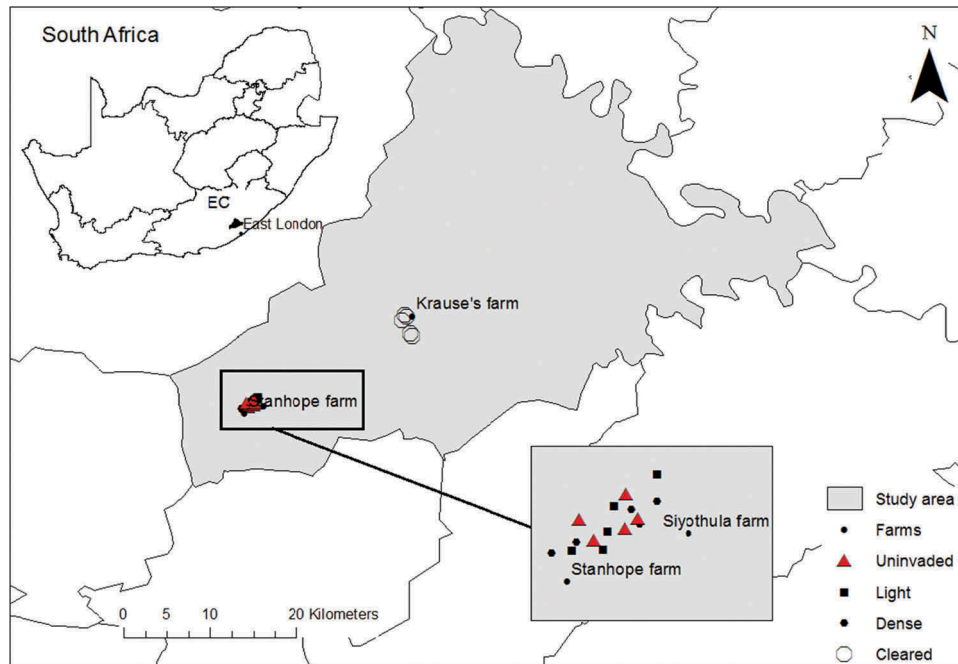


Figure 1. Map showing location of the study site and the placement of sampling plots.

uninvaded sites we compared the rangeland condition score and forage potential (Trollope et al. 2006) of the uninvaded sites to that of the benchmark sites (Trollope 1986). The rangeland condition score of the uninvaded sites was slightly lower than that of the benchmark, but with similar forage potential. Cleared sites were selected to measure recovery of grassland ecosystem following removal of *A. mearnsii*. All five replicates of cleared sites were on one farm as there were no other cleared sites in reasonable proximity. We acknowledge that we do not have true replicates of cleared sites, but it does not negate undertaking such work, but it does demand some caution in interpretation of the results. To minimise the effects of pseudo-replication, sample sites were widely spread out on a large area. The limited number and size of the farms available and the need to space out sample plots restricted our sample size. Although small, some of the key variables had low variability and consequently some statistically clear and interesting results were obtained. In these instances, a larger sample size is unlikely to have changed the outcome. However, a large sample may have provided greater power in testing of treatments differences for variables with high variability. Field surveys were undertaken during the period December 2011 to February 2012. The initial clearing of *A. mearnsii* was done 5 years prior to assessment (cleared in 2006). Discussions with landowner confirmed the state of tree density prior to clearing as equivalent to that of densely invaded sites. Clearing operations involved the complete mechanical removal of *A. mearnsii* by cutting and the application of herbicide on cut

stumps to prevent resprouting. Follow-up treatments were applied every year after the initial clearing, and sites were burnt after clearing to stimulate germination from the soil stored seed bank.

2.2.1. Tree cover

Tree density was determined using the point-centred quarter method (Cottam and Curtis 1956), as it is a more suitable method to physically sample the dense wattle stand. The point-centred quarter method collects information on density, frequency, coverage if tree species and only requires a small sampling across the woodland instead of measuring all present trees (Cottam and Curtis 1956). One transect of 100 m in the centre of each plot was used to record points at 10 m intervals along each transect. The distance to the nearest tree in each of the four quarters surrounding a recording point was measured and the tree (in case of a multitemmed plant, distance was measured to the stem located at the centre). There were no other tree species present in the sample plots other than *A. mearnsii*.

Mean distance between wattle tree stems was compared between lightly and densely invaded sites. Subsequently, we defined light invasion as greater than 3 m mean distance and dense invasion as less than 3 m mean distance. Mean distance between wattle trees was used to estimate tree density (plants ha⁻¹) within each density. Densities were calculated by dividing the square of the mean distance (D), expressed in m², into the area of hectare (10 000 m²/D²) (Cottam and Curtis 1956).

2.2.2. Vegetation sampling

At each site, three 100 m parallel transects were laid out 10 m apart. At first, transect was selected in the

centre of the site and two other transects were located 10 m on each side. Botanical composition and the basal cover of the herbaceous layer were assessed using the point-to-tuft method described by Trollope (1986). Along each 100 m transect, 50 recordings of species composition were made at 2 m intervals. At each 2 m interval one herbaceous species at or nearest to the point within a 30 cm radius was identified and recorded. If there was no vegetation within the 30 cm radius, the point was recorded as bare ground, to determine basal cover. Grass species were grouped as being Decreaser and Increaser species according to their response to grazing (van Oudtshoorn 2006). This ecological grouping system, described below, was then used to determine rangeland condition at each site (van Oudtshoorn 2006). Decreaser grass species are species that dominate in rangelands that are optimally utilised, but decrease in relative abundance in rangelands that are over- or underutilised. Increaser species are indicators of the poor range condition and are divided into 3 classes: Increaser I, II and III. Increaser I species are generally unpalatable climax grasses that are abundant in underutilised rangeland and under conditions of little or no herbivory. Increaser II species are grasses that dominate in over utilised rangeland. These include pioneer and sub-climax species that produce much viable seed and can thus quickly establish on new exposed ground. Increaser III species increase in abundance with selective grazing, are generally unpalatable dense climax grasses that are common in over-grazed rangeland. Herbaceous species that did not fall in the above categories (including alien grasses, sedges and forbs) were listed as 'other'.

2.2.3. Soil sampling

The same plots and 100 m transects that were used for the vegetation assessment were used to survey soils. Ten 1 m² quadrats were laid out at 10 m intervals along the transects. Visual soil surface assessment was conducted in each quadrat using a method derived in part from the Landscape Functional Analysis method developed by Tongway and Hindley (2004). This method uses several indicators to assess the biogeochemical function of the landscape. The selected indicators for this study include plant litter, deposited material and soil surface roughness. Plant litter accumulation is strongly related to elements stored in the surface soil layers such as carbon and nitrogen. Plant litter was therefore assessed as an indicator for decomposition rates (O'Farrell et al. 2009). Deposited material was assessed to determine the amount of alluvium transported to and deposited in the query zone (i.e. the area being examined). Soil surface roughness is related to the ability of the surface to capture mobile resources such as water, propagules, topsoil and

organic matter therefore increasing nutrient cycling and water infiltration rate. Indicators of surface roughness such as microtopography (depressions) and high grass plant density were used to assess surface roughness (Tongway and Hindley 2004). The amount of dung found in each quadrat was recorded as a proxy for grazing intensity and herbivore pressure. Dung deposits which fell within each quadrat were recorded, and each quadrat was given a score of 1 to 5 based on the amount of dung present. Average dung deposits were calculated for each transect and summed to give that particular plot a score (Hodgins and Rogers 1997; Barnes 2001). Water infiltration rate was measured in each quadrat to assess the soil's ability to partition rainfall into soil water and runoff. A single ring infiltrometer was used to assess water infiltration rate by recording the amount of time taken for 25 ml of water to infiltrate the soil surface.

Within each quadrat, a soil sample was taken to a depth of 10 cm. Soil samples for each transect were grouped, providing three bulked samples for a mean value per site. Samples were sent to a commercial laboratory (BemLab, Somerset West), for moisture content and chemical analysis. Chemical dispersion was done using sodium hexametaphosphate (calgon). The soil was air dried, sieved through a 2 mm sieve and analysed for pH (1.0 M KCl), P (Bray II) and total extractable cations, namely potassium (K), calcium (Ca), magnesium (Mg) and sodium (Na) (extracted at pH = 7 with 0.2 M ammonium acetate) and organic Carbon (C) by means of the Walkley-Black method. The extracted solutions were analysed with a Varian ICP-OES optical emission spectrometer (The Non-affiliated Soil Analyses Work Committee 1990). Total nitrogen (N) content of soil was determined through total combustion using a Leco Truspec® CN Nitrogen Analyzer. The soil's Cation Exchange Capacity (CEC) was determined using 0.2 M ammonium acetate method as described by The Non-affiliated Soil Analyses Work Committee (1990), where after the soil was leached with 0.2 M K₂SO₄. The total NH₄⁺ was then determined using a Varian ICP-OES optical emission spectrometer as indication of CEC.

2.3. Rangeland condition scores and grazing capacity assessment

Rangeland condition refers to the state of the natural vegetation in relation to its ability to withstand soil erosion, support species diversity, provide fuel to support fire and its long-term potential to produce forage for sustained livestock production (Tainton 1999). Rangeland condition scores (RCS) were determined by multiplying the percentage cover of each species from each treatment with its forage factor, which is an index of the forage production potential for each

species (Trollope 1986). Forage factors range from 0 (low potential) to 10 (high potential), (Trollope 1986). Rangeland condition scores for species grouped in a given ecological group were summed to give that ecological group's contribution per treatment and the sums of all the ecological groups were added to give the rangeland condition score for each treatment. Ecological groups whose rangeland condition scores differed significantly between treatments were examined to establish which species were most affected and therefore responsible for changes in grazing capacities following *A. mearnsii* invasion and clearing. Grazing capacity refers to the potential of a grazeable homogeneous unit of vegetation, expressed as area of land required to maintain a single livestock unit over an extended period of time without deterioration (Tainton 1999).

Current grazing capacities (CGC) (potential of the vegetation in its current state to provide forage for livestock) for each treatment were determined using the estimates of rangeland condition scores (RCS) obtained for each treatment, as presented by Danckwerts (1989). Rangeland condition scores obtained from different treatments were each divided by the condition score of the benchmark (CSB), as determined by Trollope (1986). To drive current grazing capacity for each treatment in hectares per large stock unit (ha/LSU) the proportions obtained were multiplied by the potential grazing capacity of the benchmark (PGC) (Trollope 1986) to derive current grazing capacity for each treatment in hectares per large stock unit (ha/LSU). The formula was:

$$\text{CGC} = \text{PGC} \times (1/(\text{RCS}/\text{CSB})) \text{ (Danckwerts 1989).}$$

2.4. Statistical analysis

This study tested the impacts of *A. mearnsii* invasion on grazing capacity of montane grasslands of the Eastern Cape. Four treatments (uninvaded, lightly invaded, densely invaded and cleared) were applied to compare the impacts of *A. mearnsii* invasion on the mean grazing capacities of montane grasslands. Mean grazing capacities were compared between treatments (uninvaded, cleared, lightly invaded and densely invaded). Impacts of *A. mearnsii* invasion and clearing on soil physical properties (infiltration rate and moisture content), soil surface properties (litter cover, deposited material, surface roughness and dung density) and chemical properties (pH, P, N, C, Na, K⁺, Ca⁺, Mg⁺ and CEC) were evaluated by comparing mean concentrations in different treatments. Significant differences in the above variables between the treatments were taken to represent the impacts of invasion and clearing of *A. mearnsii*. Standard errors were used to express variation around

means. The effects of different treatments on means were compared using a one way ANOVA in STATISTICA 12 (StatSoft Inc 2012). Data were first tested for normality using Kruskal-Wallis. Effects of different treatments on means were compared using Welch ANOVAs followed by Games-Howell post-Hoc test, as the data was not normally distributed. Differences between means were considered significant at $p \leq 0.05$.

3. Results

3.1. Vegetation assessment

3.1.1. Tree density

Mean distance between tree stems on lightly and densely invaded sites was 4.0 and 2.7 m, respectively (Cottam and Curtis 1956). Densely invaded sites had higher tree density (1 371.74 plants ha⁻¹). Density was much lower in lightly invaded sites (625 plants ha⁻¹).

3.1.2. Impacts of *A. mearnsii* invasion and clearing on individual species basal cover

The mean distance between wattle stems on densely invaded sites was significantly shorter ($F = 4.467$, $p < 0.001$) than on lightly invaded sites. Invasion by *A. mearnsii* reduced species basal cover by 15 and 42% on lightly and densely invaded sites, respectively, when compared to uninvaded sites. There was no significant difference between the mean basal cover on uninvaded and lightly invaded sites. Mean herbaceous cover was improved by clearing of *A. mearnsii* and both cleared and uninvaded sites had higher ($F = 6.62$, $p < 0.000$) basal cover than densely invaded sites. The uninvaded and cleared sites had no bare areas recorded during the study. Bare area constituted 15% of the lightly invaded site area and nearly 60% of the area of the densely invaded sites (Figure 2).

Mean percent basal cover for Decreaser species, Increaser II species, Increaser III species and species falling into the 'other' group (Figure 2) differed significantly between treatments. Mean percent basal cover of Decreaser species on uninvaded sites was higher than that on densely invaded sites. Increaser II species mean percent basal cover on cleared sites was higher than that on densely invaded sites. Increaser III species mean percent basal cover was higher on cleared sites when compared to all other categories. Cover of 'other' was higher on lightly invaded sites when compared to cleared sites (Figure 2).

3.2. Soil assessment

3.2.1. Water infiltration and moisture content

Water infiltration rate was significantly faster on lightly and densely invaded sites when compared to both uninvaded and cleared sites (Table 1). There was

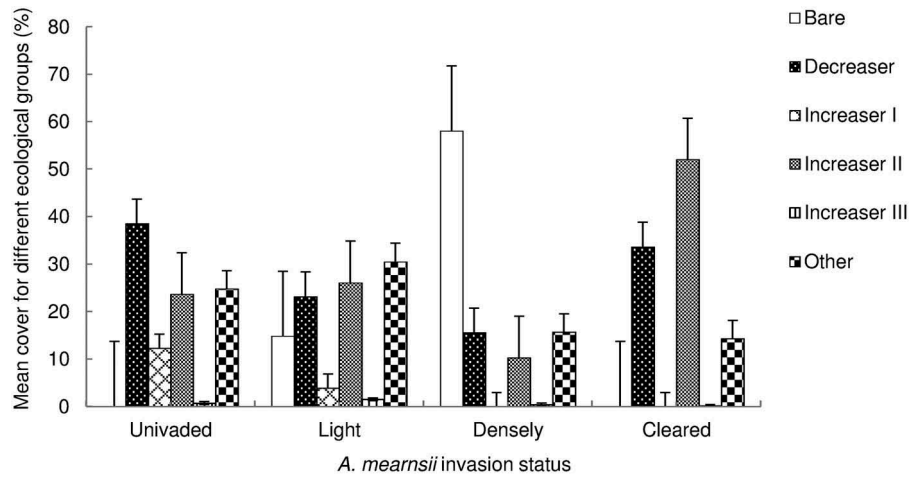


Figure 2. Mean percent basal cover \pm SE for ecological groups in uninvaded ($n = 5$), lightly invaded, ($n = 5$), dense invaded ($n = 5$) and cleared ($n = 5$) sites in the Amathole Montane Grassland of Stutterheim, Eastern Cape ($F = 6.62, p < 0.000$).

Table 1. Means \pm SE values of infiltration rate and layer properties in uninvaded ($n = 5$), lightly invaded ($n = 5$), dense invaded ($n = 5$) sites and cleared ($n = 5$) in the Amathole Montane Grassland of Stutterheim, Eastern Cape.

Treatment	Uninvaded	Lightly Invaded	Densely Invaded	Cleared	F statistic	p <
Infiltration rate (sec)	664.5 \pm 20.44 ^a	314.7 \pm 20.44 ^b	231.9 \pm 20.44 ^b	888.9 \pm 20.44 ^c	224.96	0.00
Litter	1.3 \pm 0.04 ^a	2.7 \pm 0.13 ^b	4.2 \pm 0.10 ^c	1.4 \pm 0.09 ^a	204.73	0.01
Deposited material*	4.0 \pm 0.00 ^a	3.2 \pm 0.07 ^b	2.7 \pm 0.06 ^c	4.0 \pm 0.00 ^a	170.60	0.01
Soil surface roughness	3.0 \pm 0.00 ^a	3.2 \pm 0.12 ^a	1.9 \pm 0.09 ^b	3.0 \pm 0.01 ^a	62.209	0.01
Dung density	0.8 \pm 0.08 ^a	1.1 \pm 0.19 ^a	0.3 \pm 0.06 ^b	1.1 \pm 0.10 ^a	10.137	0.01

*Following the LFA method, for deposited material the lower values indicate higher amount of deposited material and vice versa. Different letters in a row denote significant values at $p < 0.05$ (Games Howell Post-Hoc test).

no significant difference between lightly and densely invaded sites. Water infiltration rate was faster on uninvaded sites when compared to cleared sites (Table 1). There was no significant difference in soil moisture content between uninvaded and lightly invaded sites and between uninvaded and densely invaded sites (Figure 3). However, percentage soil moisture was significantly higher on lightly invaded sites than densely invaded sites. Soil moisture content was higher on the cleared sites than on densely invaded sites (Figure 3).

3.2.2. Impacts of *A. mearnsii* invasion and clearing on soil surface properties

Plant litter cover increased with increasing density of *A. mearnsii* (Table 1). Litter cover was higher on densely invaded sites when compared to lightly invaded, uninvaded sites and cleared sites. Lightly invaded sites had higher litter cover than uninvaded and cleared sites. There was no significant difference between cleared and uninvaded sites (Table 1). There was a significant difference between densely invaded sites when compared to uninvaded and cleared sites

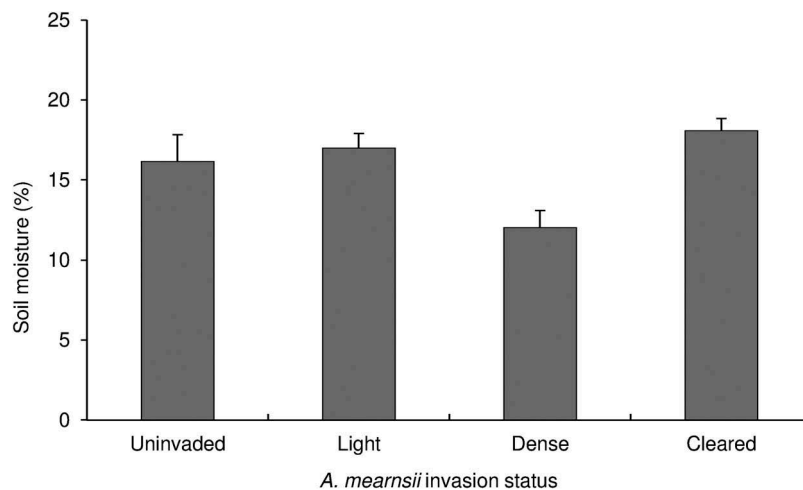


Figure 3. Average percent moisture content \pm SE for difference sites: for uninvaded ($n = 5$), lightly invaded, ($n = 5$), densely invaded ($n = 5$) and cleared ($n = 5$) sites in the Amathole Montane Grassland of Stutterheim, Eastern Cape. All differences are significant at $p < 0.05$, Games Howell Post-Hoc test ($F = 5.40, p < 0.01$).

(Table 1). There was no evidence of deposited material in uninvaded and cleared sites. Deposited material was more abundant on densely invaded sites than in lightly invaded sites. Soil surface roughness was less in dense invaded sites than on uninvaded, lightly invaded and cleared sites. Dung density was lowest in densely invaded sites when compared to uninvaded, lightly invaded and cleared sites (Table 1).

3.2.3. Impacts of *A. mearnsii* invasion and clearing on soil chemical properties

Sites that were densely invaded by *A. mearnsii* were slightly more acidic compared to the uninvaded and cleared sites. There was no significant difference on pH levels between densely and lightly invaded sites and between cleared and uninvaded sites (Table 2). Cleared sites had higher pH than densely invaded sites. Densely invaded sites had higher concentrations of P, C and N than uninvaded sites (Table 2), whereas, Na and Mg showed a relative decrease in densely invaded sites compared to uninvaded sites. There were no significant differences on pH, P, C, K, Ca and CEC concentrations between lightly invaded and uninvaded sites, lightly and densely invaded and lightly and cleared sites (Table 2). However, differences were noted for N which increases significantly during light invasion, as well as for Na which decreases significantly when compared to uninvaded sites. Cleared sites had higher P concentrations than uninvaded sites and higher K concentrations when compared to densely invaded, lightly invaded and uninvaded sites, whereas Na concentrations were significantly lower under cleared sites when compared to uninvaded sites. Lower concentrations of Mg were noted under cleared sites when compared to uninvaded and lightly invaded sites (Table 2).

3.3. Grazing capacity assessment and species condition scores underlying changes in grazing capacity

Invasion by *A. mearnsii* reduced grazing capacity by 56% and 72% on light and densely invaded sites, respectively, whereas grazing capacity on cleared

sites had increased to levels resembling uninvaded sites (Figure 4).

Means for all grass ecological groups except for species falling into the 'other' group ($F = 4.32$, $p < 0.05$) differed significantly in rangeland condition scores between treatments. Of the four Decreaser grass species present in the study sites (Table 3), only *Themeda triandra* had a significant contribution to the higher rangeland condition score in uninvaded sites when compared to lightly and densely invaded sites. Increaser II species contributed significantly ($F = 4.32$, $p < 0.05$) to the mean grazing capacity of the cleared sites, with *Cynodon dactylon* showing significant contributions.

4. Discussion

4.1. Effects of *A. mearnsii* invasion on rangeland condition and grazing capacity

The results demonstrate a severe decline in grazing capacity as a result of *A. mearnsii* invasion. The reduction in grazing capacity, from 2 to 8 ha required to support one large livestock unit on uninvaded and densely invaded sites respectively has important socio-economic and human wellbeing implications (Stafford et al. 2016). *Acacia* species have been reported to affect conditions that are essential for grazing provision, including loss of species diversity, herbaceous cover, altered soil conditions (Ndou and Ruwanda 2016). In this study the loss of grazing capacity in densely invaded sites is highly attributed to displacement of total basal cover. The impact on lightly invaded sites include loss of grass cover and that of high grazing value Decreaser species *Themeda triandra* and *Sporobolus fimbriatus*, which were more prominent in uninvaded sites. The reduction in herbaceous and grass cover resulting from invasion by *Acacia* has also been reported by several studies (Holmes 2005; Blanchard and Holmes 2008; Morris et al. 2008; Ruwanda et al. 2013). *Acacia* species have qualities, both above and below ground, which give them a competitive advantage over native species for growth resources. The high leaf biomass results in shading of below canopy species from essential

Table 2. Mean percent content \pm SE for soil chemical properties in uninvaded ($n = 5$), cleared ($n = 5$), lightly invaded ($n = 5$) and dense invaded ($n = 5$) sites in the Amathola Montane Grassland of the Eastern Cape.

Chemical properties	Uninvaded	Lightly Invaded	Densely Invaded	Cleared	F statistic	p <
pH(KCl)	4.4 \pm 0.04 ^a	4.3 \pm 0.05 ^{ab}	4.2 \pm 0.04 ^b	4.4 \pm 0.06 ^a	6.60	0.05
P (mg/kg)	2.0 \pm 0.20 ^{ab}	5.2 \pm 1.78 ^{bc}	4.2 \pm 0.59 ^c	7.3 \pm 1.57 ^c	3.19	0.05
C%	2.0 \pm 0.10 ^a	3.0 \pm 0.41 ^{ab}	4.0 \pm 0.51 ^b	3.2 \pm 0.46 ^{ab}	4.03	0.05
N%	0.1 \pm 0.01 ^a	0.2 \pm 0.02 ^b	0.2 \pm 0.01 ^b	0.2 \pm 0.02 ^b	7.32	0.01
Na%	5.1 \pm 0.18 ^a	4.0 \pm 0.18 ^b	3.4 \pm 0.14 ^b	3.7 \pm 0.25 ^b	14.90	0.01
K%	4.1 \pm 0.22 ^a	3.9 \pm 0.20 ^a	4.2 \pm 0.12 ^a	5.7 \pm 0.37 ^b	10.87	0.01
Ca%	33.4 \pm 1.53	35.3 \pm 1.31	36.3 \pm 0.82	35.9 \pm 1.25	1.06	ns
Mg%	22.8 \pm 0.63 ^a	21.0 \pm 0.68 ^{ab}	19.6 \pm 0.66 ^{bc}	17.0 \pm 0.94 ^c	11.05	0.01
CEC(cmol (+))/kg	8.0 \pm 0.79	10.7 \pm 0.79	10.8 \pm 0.53	8.6 \pm 0.58	17.11	Ns

Different superscripts in a row denote significant values at $p < 0.05$ (Games Howell Post-Hoc test).

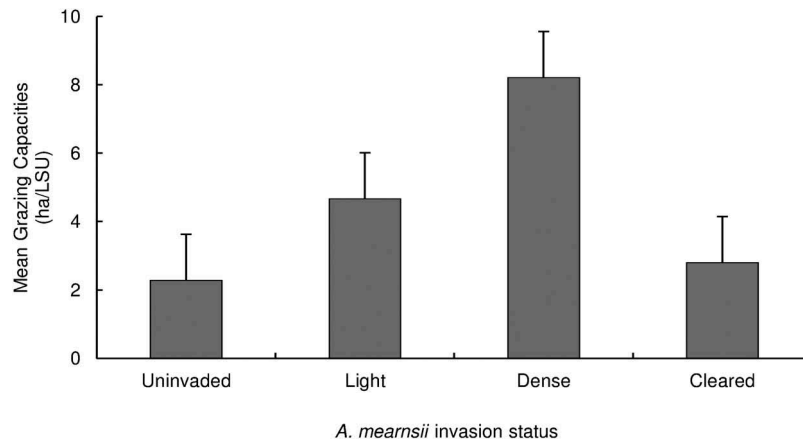


Figure 4. Mean grazing capacities \pm SE for uninvaded ($n = 5$), lightly invaded, ($n = 5$), densely invaded ($n = 5$) and cleared ($n = 5$) sites in the Amathole Montane Grassland. All differences are significant at $p < 0.05$, Games Howell Post-Hoc test ($F = 14.84$, $p < 0.05$).

Table 3. Mean range condition scores \pm SE for different species in uninvaded ($n = 5$), lightly invaded, ($n = 5$), densely invaded ($n = 5$) and cleared ($n = 5$) sites in the Amathole Montane Grassland of Stutterheim, Eastern Cape.

Species	Uninvaded	Lightly Invaded	Densely Invaded	Cleared	F statistic	p <
Decreasers						
<i>Cenchrus incertus</i>	2.4 \pm 1.48	1.1 \pm 0.7	0.6 \pm 0.64	0.0 \pm 0.00	3.77	ns
<i>Panicum maximum</i>	0.0 \pm 0.00	2.9 \pm 2.93	3.7 \pm 2.78	0.0 \pm 0.00	3.77	ns
<i>Sporobolus fimbriatus</i>	31.6 \pm 9.65	28.6 \pm 6.91	18.5 \pm 6.17	44.8 \pm 2.62	3.77	ns
<i>Themeda triandra</i>	28.0 \pm 5.80 ^a	0.8 \pm 0.80 ^b	0.0 \pm 0.00 ^c	3.2 \pm 2.33 ^{bc}	3.77	0.01
Increaser I						
<i>Cymbopogon plurinodis</i>	0.1 \pm 0.11	0.0 \pm 0.00	0.0 \pm 0.00	0.0 \pm 0.00	15.62	ns
<i>Hyparrhenia hirta</i>	0.2 \pm 0.21	0.0 \pm 0.00	0.0 \pm 0.00	0.0 \pm 0.00	15.62	ns
<i>Miscanthus capensis</i>	0.0 \pm 0.00	1.6 \pm 1.55	0.0 \pm 0.00	0.0 \pm 0.00	15.62	ns
<i>Tristachya leucothrix</i>	16.6 \pm 6.87	0.0 \pm 0.00	0.0 \pm 0.00	0.0 \pm 0.00	15.62	ns
Increaser II						
<i>Cynodon dactylon</i>	0.3 \pm 0.26 ^a	2.5 \pm 0.47 ^{ab}	2.0 \pm 0.34 ^{ab}	10.9 \pm 2.35 ^b	13.82	0.00
<i>Eragrostis capensis</i>	2.3 \pm 2.13	3.4 \pm 3.02	0.1 \pm 0.11	0.9 \pm 0.55	13.82	ns
<i>Eragrostis chloromelas</i>	0.0 \pm 0.00	0.0 \pm 0.00	0.0 \pm 0.00	0.9 \pm 0.43	13.82	ns
<i>Eragrostis racemosa</i>	0.9 \pm 0.78	3.3 \pm 1.74	0.3 \pm 0.20	0.0 \pm 0.00	13.82	ns
<i>Eragrostis trichophora</i>	0.0 \pm 0.00	0.0 \pm 0.00	0.0 \pm 0.00	0.1 \pm 0.11	13.82	ns
<i>Eragrostis viscosa</i>	0.0 \pm 0.00	0.0 \pm 0.00	0.0 \pm 0.00	0.3 \pm 0.32	13.82	ns
<i>Panicum schinzii</i>	0.0 \pm 0.00	0.0 \pm 0.00	0.0 \pm 0.00	0.0 \pm 0.00	13.82	ns
<i>Sporobolus africanus</i>	25.2 \pm 4.38	10.6 \pm 4.47	6.0 \pm 3.50	27.4 \pm 7.33	13.82	ns
Increaser III						
<i>Merxmuellera stricta</i>	0.4 \pm 0.40	0.9 \pm 0.78	0.2 \pm 0.16	0.1 \pm 0.08	4.06	ns
Other						
<i>Cyperus rotundus</i>	0.0 \pm 0.00	0.0 \pm 0.00	0.0 \pm 0.00	0.0 \pm 0.00	3.60	ns
<i>Forb</i>	0.0 \pm 0.00	0.0 \pm 0.00	0.0 \pm 0.00	0.0 \pm 0.00	3.60	ns
<i>Kyllinga erecta</i>	0.0 \pm 0.00	0.0 \pm 0.00	0.0 \pm 0.00	0.0 \pm 0.00	3.60	ns
<i>Paspalum dilatatum</i>	0.4 \pm 0.43 ^{ab}	0.5 \pm 0.24 ^{ab}	0.0 \pm 0.00 ^b	4.8 \pm 1.30 ^a	3.60	0.05
<i>Paspalum notatum</i>	6.9 \pm 3.14 ^a	0.0 \pm 0.00 ^b	0.3 \pm 0.32 ^{ab}	0.0 \pm 0.00 ^b	3.60	0.05

Superscript values denote significant differences at $p < 0.05$ (Games Howell Post-Hoc test).

growth resources such as light, reducing the amount of precipitation that reaches the ground (Everson et al. 2014). As a consequence, *Acacia* biomass can alter habitat suitability for indigenous vegetation by reducing light penetrability which changes the microclimate below the canopy tree canopy (Holmes and Cowling 1997; Morris et al. 2011). Furthermore, high leaf litter content deposited under tree canopy, as evident in the densely and lightly invaded sites in this study, may change the nutrient status of the soil to be more suitable for the invading species. The high root biomass of *A. mearnsii* is associated with high moisture and nutrient uptake (Dye and Jarman 2004; Yelenik et al. 2007; Morris et al. 2011).

In this study, dense invasion reduced soil moisture by 4.97%. Because of the reported comparatively excessive water use by *Acacia* species (Dye and Jarman 2004; Gaertner et al. 2011; Le Maitre et al. 2016), we anticipated that all invaded sites would have significantly less soil moisture. However, local conditions and factors such as rainfall during the survey period played a more influencing role. The findings of studies such like the one by Yelenik et al. (2004) confirm this, finding higher soil moisture content under dense *Acacia* stands compared with their fynbos reference sites. Above average rainfall during the survey period for this study may have resulted in higher soil moisture in uninvaded and

cleared sites. Water infiltration rate was higher in lightly and densely invaded sites than in uninvaded and cleared sites. High water infiltration rates on invaded sites could be attributed to degraded soil surface conditions and poor basal cover; this is in line with the findings by O'Farrell et al. (2009). This could imply that while infiltration rate is faster on invaded sites, the soil may also dry out faster. Whereas in uninvaded sites, good cover of standing biomass slows water runoff and allows time for water to infiltrate into the soil. Similarly, Mills and Fey (2004) in their study, associated water infiltration with high crust brokenness in invaded sites. Furthermore, trampling on bare soil by grazing animals can affect soil structure, and therefore water infiltration rates, and surface water runoff (Pietola et al. 2005; Eldridge and Whitford 2009; Dunne et al. 2011), particularly in lightly invaded sites as livestock tend to congregate here during extreme weather conditions. As anticipated, we found higher litter content in invaded sites when compared to uninvaded and cleared, with densely invaded sites having almost double the amount of plant litter on lightly invaded sites. This can be explained by the high leaf biomass produced by *A. mearnsii* as found by other studies (Yelenik et al. 2007; Gaertner et al. 2011; Witkowski 2012). High soil surface roughness on uninvaded sites was explained by presence of standing grass tussocks. The lower dung content found under densely invaded sites is an indication of low grazing animal activity (Hodgins and Rogers 1997; Barnes 2001). Indicating that the dense wattle stands can form a barrier to animal movement (De Neergaard et al. 2005). Low grass cover at these sites further supports the notion that these environments are avoided by grazing animals. The presence of some palatable grass species in the lightly invaded sites and higher dung content of these sites suggests that these sites are being utilised to a higher degree by animals than densely invaded sites. *Acacia* trees at this density do not create a barrier to animal movement, unlike densely invaded sites (De Neergaard et al. 2005), and may provide shade and shelter during extreme weather conditions.

The consequences of *A. mearnsii* invasion on soil nutrient status were more evident in the densely invaded sites than lightly invaded sites. Findings of this study concur with those of Montgomery (2001), where densely invaded sites had more acidic soils than uninvaded sites. Previous studies (Yelenik et al. 2004; Witkowski 2012) have found that *A. mearnsii* invaded sites had higher concentrations of P, C and N. In contrast (Oelofse et al. 2016), C content (at 60 cm) in the grassland of Matatiele in Motseng area was reported to be higher when compared to wattle invaded sites. In the same study, Oelofse et al. (2016) found no differences in C content between grassland

and wattle invaded sites in the Madlangala study sites, in the same grassland type. They concluded that the lack of differences between grassland and wattle stand in Madlangala could be due to management effects and burning. Yelenik et al. (2007) found no differences in K and Na concentrations under *A. saligna* canopy but reported higher levels of Mg and Ca when compared to the fynbos species dominated soils. Results of this study showed low levels of Mg and Na concentrations under *A. mearnsii* canopy, whereas there were no differences in Ca and K concentrations under all invaded treatments. In this study, the higher the concentration of P, C and N on invaded sites could be attributed to a combination of factors, including addition of these elements from nutrient rich plant litter biomass from wattle trees. This may also be an effect of the relatively short time period since clearing. Witkowski (1991) and Yelenik et al. (2007) associated higher levels of nutrients with the redistribution of soil nutrients by *Acacia* trees from deeper layers of the soil. *Acacia* species have been associated with high nutrient uptake when compared to native vegetation, and this might be the cause of low concentrations of some nutrient levels under *A. mearnsii* canopy in this study. Increased soil N concentrations under *Acacia* canopy have been reported by several studies and have been associated with high levels of nitrogen-rich plant litter. *A. mearnsii* is a leguminous tree with N₂ fixing capability (Yelenik et al. 2004) and has the capacity to increase soil N concentrations.

4.2. Changes in rangeland condition and grazing capacity following the clearing of *A. mearnsii*

The results showed a considerable increase in rangeland condition and gains in grazing capacity 5 years after the clearing of *A. mearnsii*. These improvements could be attributed to a general increased cover of the grass layer since the removal of *A. mearnsii*. The increased abundance of Decreaser and Increaser II species on cleared sites indicating an increase in palatability resulted in high grazing index values (Trollope 1986; Van Oudtshoorn 2006). The improvement of herbaceous vegetation cover after *Acacia* clearing has been reported by several studies (Blanchard and Holmes 2008; Morris et al. 2008; Ruwanza et al. 2013). Clearing of *A. mearnsii* would have released the pressure on soil nutrients and other essential resources such moisture and light and enhanced availability of these growth resources to allow plant successional pathways to take place (Yelenik et al. 2004). Morris et al. (2008) and Ndhlovu et al. (2011) reported an increase in species richness, cover and abundance of native vegetation after complete removal of alien invasive species. They

found that proper follow-up treatments after clearing contributed significantly to the recovery of indigenous vegetation.

High rainfall during our field survey period may have masked any longer-term effects of invasion on soil moisture. However, our results show that soils in cleared sites could hold soil moisture much longer than densely invaded sites. As reported by Everson et al. (2014) dense wattle invasion could have intercepted much of the rainfall resulting in low soil moisture in densely invaded sites. Furthermore, high moisture content and therefore slow water infiltration rates on cleared sites could be attributed to good herbaceous cover. Clearing of trees reduces water loss from the soil by wattle trees. The absence of the wattle canopy also allows more precipitation to reach the soil surface and therefore, improves soil water availability to the grass species (Everson et al. 2014).

Plant litter was significantly lower on cleared sites than on invaded sites, and this could be explained by the absence of litter-producing *Acacia* species (Yelenik et al. 2004; Witkowski 2012). Furthermore, the absence of deposited material in both uninvaded and cleared sites could be attributed to good standing grass tussock and the absence of *A. mearnsii*, respectively (Mills and Fey 2004).

The legacy effects of *A. mearnsii* appear to have raised the soil fertility of cleared sites. Concentrations of soil nutrients on cleared sites were similar to that of densely invaded sites as has been reported by previous studies (Yelenik et al. 2004, 2007; Gaertner et al. 2011). In this study, the short period since clearing is likely to have contributed to the observed similarities between densely invaded and cleared sites. Furthermore, perhaps soil samples taken at a more shallow depth (e.g. 1–3 cm) would likely have provided more information on the soil nutrients. Soil pH and K concentrations on cleared sites were higher than on densely invaded sites and similar to that of uninvaded sites (Montgomery 2001). Soil phosphorus, organic carbon and nitrogen content were the same on invaded and cleared treatments. Corbin and D'Antonio (2004) suggested that *Acacia spp.* have an extended legacy after clearing and this result in a lag effect with the slow release of available N from the higher N pool in the soil. Furthermore, high grass cover on cleared sites is likely to have contributed to increased soil nutrients on cleared sites by reducing water run-off, thus the loss of organic matter and nutrients.

Our study clearly demonstrates that significant grazing resources are lost under conditions of *A. mearnsii* invasion. Clearing of *A. mearnsii* can help to restore the decline in grazing capacity, but this is conditional on the autogenic recovery of the grass layer. Increased soil nutrient levels, which can remain for an extended period after clearing of *A. mearnsii*,

could have the impact of improving grazing condition. However, considering that all cleared sites were located on the same farm, there is no way of separating the role of management effects on the successful recovery of cleared sites. We speculate that the successful recovery following alien removal could be due to well-timed-follow up treatments. Wattle seed bank sizes, disturbances during clearing, and the potential abundance of nutrients increase the potential for re-establishment of the same species, therefore careful monitoring of cleared sites is essential to ensure ecosystem recovery. The costs involved in clearing alien invasive plants is very high (Marais et al. 2004) and it is unrealistic to assume that poorly resourced, emerging farmers will be able to undertake such interventions. Therefore, more research is needed to understand where in the landscape restoration is mandatory and where follow-up alone is needed.

The overall implications here are that the negative impacts of invasive alien plants on grazing resource as an ecosystem services that supports livestock production and associated livelihoods are substantial and warrant the same focus that impacts of invasive alien plants on water resources have received. Furthermore, if alien plant invasions continue unabated, South Africa's livestock production and therefore agricultural economy will most certainly be threatened by invasive alien plants. The cost of wattle clearing can be prohibitive so government interventions that assist emerging farmers in this process of clearing invasive alien plants will both enhance the speed of recovery and the chances of creating sustainable rural economies.

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