Range degradation and land tenure change: insights from a 'released' communal area of Eastern Cape province, South Africa

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RANGE DEGRADATION AND LAND REFORM: INSIGHTS FROM A 'NEW' COMMUNAL AREA OF EASTERN CAPE PROVINCE, SOUTH AFRICA

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ABSTRACT

In South Africa the relative extent of range degradation under freehold compared to communal tenure has been strongly debated. This study presents findings from a 'new' communal area, where a former commercial freehold farm was transferred to communal ownership in 1976 and compares this with an ecologically similar farm still under freehold tenure. Analysis of historical aerial photographs demonstrated an increase of 21.5% in shrub abundance at the communal site immediately after it changed from freehold tenure (1975-1985), but a slight decline of 0.3% in the years preceding the change (1968-1975). Conversely, at the freehold site, shrub abundance declined by 12.8% over the period 1968 to 2004. Field measurements revealed significantly ($p<0.05$) greater frequency of both woody and dwarf shrubs under communal tenure as well as significantly lower total basal vegetation cover and point to tuft distance (PTD) values for both basal vegetation and grasses. However, mean range condition score was significantly ($p = 0.03$) higher at the communal site due to the significantly ($p = 0.01$) greater frequency of *Themeda triandra*. We conclude that whilst the communal site has been able to maintain short term productivity, the absence of fire as a management tool, combined with spatially inconsistent high pressure grazing has initiated a trajectory of vegetation degradation. This has implications for current policy, suggesting that critical criteria for land transfer to historically disadvantaged groups should not only be the potential for resultant livelihood benefits but also, over the longer term, the ability of the recipients to manage the grazing resource in a way that maintains its ecological integrity.

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INTRODUCTION

Rangelands provide a vital livelihood stream to people across the globe through the range of goods and ecosystem services they deliver (Reid *et al*., 2008, Vavra and Brown, 2006). A key aspect of this is livestock production, which in semi-arid systems generally occurs on an extensive basis through either private ranching under freehold tenure or on a collective basis on land that is under communal tenure (Niamer-Fuller and Turner 1999, Reid *et al*., 2008). A major threat to the productivity of these systems is land degradation, which is estimated to cost some US$40 billion annually (FAO 2010). The current interpretation of land degradation has moved beyond the purely biophysical (vegetation change and soil loss) to be considered as '….a reduction in the capacity of land to perform ecosystem functions and services that support society and development' (FAO 2010:1). However, considerable debate still remains around the extent to which land degradation is occurring under different management and land tenure systems and the main drivers of this (Ellis and Swift, 1988, Hary *et al*., 1996, Ward *et al*., 1998, Rowntree *et al*., 2004).

Although land degradation has been identified under both freehold and communal tenure, much of the debate has focused on communal systems as these were historically perceived as the most vulnerable due to the inability of users to collectively manage commons resources on a sustainable basis – the so-called 'tragedy of the commons' (Hardin, 1968). However, subsequent empirical and theoretical developments have shown that collective action for management of common pool resources is possible (e.g. Berkes, 1989, Ostrom, 1990 and 1999) and that this can facilitate sustainable resource management by linking with ecological systems to build resilience (Berkes and Folke, 1998). Linked to this has been protracted debate regarding the degree to which rangeland change is driven by biotic or abiotic factors. Of particular focus has been the ecological dynamics of semi-arid rangelands as inherently non-equilibrial systems, which are primarily driven by abiotic factors such as rainfall, and the influence this has on vegetation dynamics in the shorter term (Behnke and Scoones, 1993). Whilst there is ample evidence to uphold the assertion that rainfall is be a key driver of rangeland vegetation dynamics, the emerging consensus is that semi-arid rangelands may exhibit a variety of equilibrial and non-equilibrial responses at different temporal and spatial scales, particularly during dry periods when feedback between plants and animals is likely to be most apparent (Illius and O'Connor 1999, Briske *et al*., 2003, Vetter 2005). Moreover, in the longer term the influence of elevated CO_2 levels as an abiotic driver of rangeland change provides more cause for concern, particularly in the possible impact this may have on increasing the proportion of trees and shrubs relative to grass (Scholes and Archer 1997, Bond *et al*., 2003).

In South Africa these debates have had particular relevance in an environment where livestock production is polarised between extensive privately-owned commercial farms, in what was historically white South Africa, and collective livestock production by black South Africans on communally held rangelands. These communal rangelands are concentrated in the former homeland areas or 'native reserves', which constitute about 13% of the land surface of South Africa but are home to nearly one quarter of its human population and hold about half of all livestock (Scogings *et al.*, 1999). These areas were first created in the late 19th Century and successively expanded until their final consolidation as 'independent' homelands during the 1970s and early 1980s. Thus, there is a varied, although in some cases considerable, history of communal grazing within these areas and associated claims of land degradation. The first official reports of land degradation in the form of overgrazing and soil erosion were recorded during the 1880s in the Herschel district of the former homeland of Ciskei (Bundy, 1988) and by the 1920s were widespread in both the Ciskei and Transkei (Bundy, 1988, Beinart, 2003). More recently a number of studies have been published demonstrating land degradation in the communal rangelands of South Africa in the form of soil erosion (e.g. Weaver, 1991, Kakembo and Rowntree, 2003, Vetter, 2007, Kakembo *et al.*, 2009), change in the composition and cover of basal vegetation (e.g. O'Connor, 1991, Kakembo, 2001, Vetter *et al*., 2006, Anderson and Hoffman 2007) and increases in woody shrubs (Roques *et al*., 2001, Shackleton and Gambiza, 2008), many involving comparison with commercial livestock farms. However, other key aspects of communal systems have received relatively little research focus. For example, few studies have attempted to relate rangeland degradation at communal sites to the amount of time they have been under communal management. In the current political environment, this will be an increasingly important consideration. The post apartheid government is committed to improving the livelihoods of formerly disadvantaged people in rural areas and a key part of this has been a focus on greater levels of land transfer and increased engagement with small-holder agriculture (Department of Agriculture, 2008). The South African Land Reform Programme has facilitated the transfer of a small but significant portion of former commercial farmland to black ownership on a communal basis as part of the restitution and redistribution process, and the pace of transfer is now beginning to increase (Lahiff, 2008). However, despite isolated studies (e.g. Vetter and Goqwana, 1999) little research is being undertaken to understand the implications this has for the continued integrity of rangeland resources. In particular published temporal studies of the impact of tenure shift on rangeland resources are lacking, possibly because few transfers have been in place for much more than 15 years.

The former Ciskei homeland in Eastern Cape Province affords an opportunity for such studies to take place. Not only have several commercial livestock farms adjoining the former homeland been recently transferred to communal ownership as part of the land redistribution process, but many commercial farms were also incorporated into the former Ciskei during the consolidation process that occurred in the late 1970s and early 1980s (Wotshela, 2001). Thus, the area currently constitutes a mosaic of commercial farms and 'new' communal settlements of varying age within a relatively small area and, in several cases, contiguous with one another. The presence of former commercial farms which have been under communal tenure for 30 years or more provides an opportunity to investigate the impact of this on rangeland quality relative to current commercial farms, within broadly similar ecotopes. Such studies may provide an indication of the types of rangeland change that can be expected on more recently acquired communal farms and thus the potential for transferred farms to deliver ecosystem functions that support communal farming in the longer term.

To this end, this study aims to build on limited research about the impact of tenure change on rangeland degradation by undertaking a detailed assessment of vegetation degradation in time and space at a site under ‘new’ communal management (transferred ownership) compared with a similar site where commercial tenure has been perpetuated. Vegetation degradation is recognised to be an important measure of rangeland degradation over different spatial and temporal scales in arid and semi-arid systems (Abel and Blakie, 1989, Behnke and Scoones, 1993). Shorter term indicators of degradation include changes in composition of basal plants, which may be reversible, but a shift in state from grasslands to systems dominated by bushes and woody shrubs is indicative of more permanent degradation (Milton and Hoffman, 1994, Dougill and Trodd, 1999). Loss of basal plant cover is also recognised to be an important precursor of largely irreversible processes such as soil erosion (Morgan, 2005). On this basis the specific objectives of the study were: -

1. To use landscape-scale measurements to determine if vegetation in similar ecotopes has degraded in the medium term under communal and freehold tenure.
2. To compare the extent of vegetation degradation currently apparent under these different tenure regimes as demonstrated by differences in species quality, basal cover and abundance of shrubs

3. To examine the variables explaining these patterns of vegetation degradation and how these differ under each tenure system.

STUDY AREA

The study sites are the former freehold farm of Allanwater, which was transferred to communal ownership in the late 1970s, and the freehold commercial farm Pink Valley. Both are located in Lukhanji Local Municipality in the northern part of the former Ciskei, and are separated by a distance of approximately 8km at their nearest points (Figure 1). The initial intention was to select sites that were in closer proximity to each other. However, it proved difficult to locate functioning commercial farms that were contiguous with settlements that had been under communal tenure since the consolidation of the former Ciskei. Rather, most of the commercial farms in this position have been abandoned and occupied by communal farmers from adjoining settlements. It was thus necessary to select a commercial farm (Pink Valley) situated on the Eastern site of the main arterial road, where commercial livestock farming continues to be practised.

INSERT FIGURE 1

Pink Valley farm is located just off the main R67 highway about 18 km South East of Sada at an elevation of 1280-1520 m a.s.l (Figure 1). Rainfall records for the period 1993-2008 give a mean annual rainfall of 547 mm (CV=0.22). According to the local classification system, the dominant rangeland types at the site are Karroid *Merxmuellera* mountain veld and Dry *Cymbopogon-Themeda* veld (Acocks, 1988). The farm, under private tenure, is 1100 ha in area, and is a mixed livestock enterprise stocked with 1300 sheep and 120 cattle. Using the standard relationship of 1 Large Stock Unit (LSU) (equivalent to a cow of 450 kg) being equivalent to 6 Small Stock Units (SSUs), the total number of equivalent large stock units at the site is 337 LSU. This equates to a mean stocking rate of

3.3ha/LSU. The farm is divided by fencing into 22 camps and the active camps are grazed on a two week rotational grazing cycle during the summer. During the dry season most of the lower camps are available to livestock simultaneously. Although fire is a recommended management tool for these veld types, particularly when encroached by karroid shrubs (Beckerling *et al*., 1995), the camps had not been burnt for about 20 years at the time of fieldwork (K.C.M. pers. comm. August 2009).

Allanwater is situated approximately 15 km south of the township of Sada, at an elevation of between 1310 and 1560 m a.s.l (Figure 1). Although long-term rainfall figures are unavailable, interpolated mean annual rainfall is 504 mm (Schulze and Lynch, 2007). The dominant rangeland types at the site are Karroid *Merxmuellera* mountain veld and Dry *Cymbopogon-Themeda* veld (Acocks, 1988), although more recent veld assessment suggests areas of Dohne Sourveld are also present (ECDA, 2002). The history of occupation and subsequent management of the Allanwater settlement is well documented. The former commercial farm was transferred to the Ciskeian government in 1976 with the intention of it becoming a government ranch for livestock improvement and extension (Wotshela, 2001). However, shortly after it was acquired it was occupied by a small group of refugees who had been displaced from the Glen Grey area of the Ciskei following its redesignation as part of the Transkei and initially settled in the overcrowded Zweledinga settlement to the north. The total land area of the settlement is ~2,500 ha, which now includes not only the original Allanwater farm but also the adjacent farms of Beaconsfield and Claremont A (Figure 1). These two properties were effectively annexed during the 1990s, as livestock holdings at the settlement expanded. The most recent figures available indicate that total livestock holdings at the site are 951 cattle, 2354 sheep and 901 goats (ECDA, 2002). This equates to 1494 equivalent LSUs in total. For the available grazing land, this gives a mean stocking rate of about 1.7ha/LSU, which is high for any veld type according to commercial recommendations. The highly dynamic nature of livestock movement in most communal systems make stocking rates notoriously difficult to assess, but for Allanwater this is probably a realistic estimate as livestock are maintained within boundary fencing (see below).

Grazing management at Allanwater has been well documented as part of previous research (Bennett *et al*., 2010). Rangeland is managed by the community under a basic common property regime with clear rules about who has access to the resource (residents of the village only) and defined resource boundaries. Boundaries and thereby grazing access are enforced through fencing most of which was inherited from the commercial farm. Efforts are made by the community to maintain this, and assistance in this respect has also recently been provided by government and the National Wool Growers Association (NWGA). A system of rotational resting and grazing has been in place since the community was established. Initially, this was under the jurisdiction of the former homeland system with decisions about opening and closing of camps being decided by a grazing committee, sanctioned by the headman and local magistrate and enforced by a resident ranger. Since the demise of the Ciskei homeland in the early 1990s, the community has perpetuated a limited form of grazing management under the control of a Common Property Association (CPA) called Vukani Farmers' Association (VFA), to which all farmers at the community belong. Grazing management involves 18 camps in total, some of which are sub-divided. Two are rested each year and a notional attempt is made to rotationally graze the remainder. However, due to lack of permanent water on the western camps and the desire of owners to graze small-stock in close proximity to the village (mainly on adjacent camps) rotational grazing in a commercial sense is not strictly practised. Rather, a limited form of seasonal grazing is practised, whereby cattle are grazed exclusively on the western camps during the summer (wet season) and then restricted to the eastern camps during the winter (dry season). Fire is not actively used as part of the management process as it is believed to limit forage available to animals and also to damage fencing.

METHODS

Assessment of the vegetation at each study site was undertaken at two different scales, involving an analysis of broader scale historical change using secondary data and a more detailed current assessment through fieldwork. We assessed the extent of degradation through time at each study site using historical landscape level data. In order to generate a basic comparison of vegetation productivity and change through time at each site, we accessed the available MODIS-LAI (Leaf Area Index) remote sensing data for each site. LAI data were available as a mean value taken over an 8 day interval for a period of 10 years (from early 2000) giving 441 data points in total. This provided a comparison of productivity between the sites and allowed determination of a basic trajectory of change in productivity at each site, albeit over a limited time period.

In order to establish if bush encroachment had taken place, aerial photographs of each study site were obtained for the years 1968, 1975, 1985, 1995 and 2004. The photographs from 1968-1995 were available in hard copy only and were digitised as high resolution files before being imported into ARC-GIS. Images from 2004 were obtained in digital format. Sections of each site, representative of the different aspects and elevations present, were purposively selected and were magnified to a standard scale of 1:10 000 across all images. This was about the maximum resolution that could be obtained before the images became too indistinct to enable adequate identification of vegetation. For the 1995 images, little could be distinguished under magnification, as the resolution of the original photographs was too low, and these images were therefore excluded from the analysis for both sites, as was the 1975 image for Pink Valley. The time series analysis was therefore undertaken on images from 1968, 1985 and 2004 for Pink Valley and 1968, 1975, 1985 and 2004 for Allanwater. A grid subdivided into squares of 150 m x 150 m was overlain on each image and the numbers of shrub clumps, large bushes and trees within each square counted for each year at each site. Total bush counts were determined for each year and percentage change between years calculated. This not only enabled a trajectory of bush encroachment to be established at each site, but also an assessment of the level of bush abundance at Allanwater before and after the switch from communal to commercial tenure in 1976. We predicted greater levels of bush encroachment at Allanwater compared to Pink Valley, particularly after 1975.

We also measured the current extent of vegetation degradation at each study site. There are several key parameters used in measuring vegetation degradation (Behnke and Scoones, 1993, Anderson and Hoffman, 2007) and we focused on: -

- Proportion of perennial grasses compared to annuals
- Quality and quantity of forage available
- Basal plant cover
- Abundance of dwarf karroid and woody shrubs.

After 30+ years of relatively unregulated, high pressure grazing at Allanwater, we hypothesised that the proportion of perennial grasses, basal plant cover and the quantity and quality of forage available would be greater at Pink Valley. Conversely, we hypothesised greater abundance of both dwarf and woody shrubs at Allanwater.

Assessment of vegetation was undertaken during August 2009. A stratified sampling strategy was adopted at each study site to try and ensure comparability of samples. A digital elevation model was used to identify sampling sites on the basis of elevation, aspect and slope. We identified four sites for each aspect, two of which were at high elevation and two at low and then ensured that one sample site from each elevation was situated on a steep (>25º) slope and one on a shallow (<25º) slope. Slopes at Pink Valley ranged between 4º and 45º and those at Allanwater between 7º and 50º. This gave a total of 16 sampling sites at each location. Sites were distributed throughout the full extent of the grazing camps at both locations. At Allanwater sample sites were located only within the boundaries of the original Allanwater farm and not within Beaconsfield or Claremont A (Figure 1), as the exact point at which communal grazing began on these latter sites is unknown. Only 15 sites were successfully sampled at Allanwater as it proved impossible to find a steep, north facing slope at low elevation, which was accessible.

Sample sites in each study area were located using a pre-programmed, hand-held GPS unit (Garmin, e-Trex H). Vegetation sampling was adapted from the simplified range condition assessment approach developed for the Eastern Cape (Beckerling *et al.*, 1995), which has been widely employed in similar studies (e.g. Vetter *et al*., 1999 and 2006). At each sample site a single 100 m transect was used to take 100 individual point measurements (one every metre). At each point strikes on either rock, bare earth or directly on basal plant species were recorded and, in the case of strikes on rock or bare earth, the rooted species within 15 cm of the point was also identified. If no species was identified within 15 cm of the point then the point was recorded as simply rock or bare earth. Grasses were identified to species level, while herbs and karroid (dwarf) shrubs were amalgamated under these life form categories. The distance from the point to the nearest rooted plant was measured. The mean point to tuft distance (PTD) provides a proxy for basal cover (Hardy and Tainton, 1993) and also provides a crude indication of vulnerability to soil erosion (Beckerling *et al.*, 1995). At each point any woody shrub species > 25 cm in height (i.e. non-karroid) occurring within 1.5 m was also noted and the distance to the base of the stem recorded.

Vegetation data were summarised in spreadsheets by frequency and % cover for all cover types (bare earth, rock, vegetation), species and life forms by transect and overall (across all transects), for each site. Cover values were determined as the number of direct strikes on a species as a proportion of the total direct strikes within each transect. PTD values were determined from all individual values for each species or life form. Sorenson's Index based on species presence/absence and Renkonen's Index (% occurrence using cover values) was used to compare the similarity of the basal vegetation at each study site. Veld (range) condition scores (VCS) were calculated using standard local methods based on the frequency of basal species (Beckerling *et al*., 1995). Abiotic variables at each site were normally distributed and compared using *t*-tests. Biotic variables were largely not and therefore differences were statistically determined using Mann-Whitney tests. For basal vegetation, differences in frequency, cover and PTD were determined for the different life form categories as well as for two key grass species, *Themeda triandra* and *Elionurus muticus*, which were purposively selected as indicators of good and poor grazing respectively. For shrubs, differences in relative frequency and PTD were determined.

Potential abiotic drivers of differences in vegetation between the two sites were investigated using backward, stepwise multiple regression analysis (Zar, 1999). The vegetation parameters included in the analysis were VCS, PTD, basal cover, frequency of shrubs and frequency of karroid shrubs. These were regressed against the abiotic variables distance from settlement, elevation, aspect and slope, which were extracted from the digital elevation model. All analyses and statistical tests were undertaken in PASW (formerly SPSS) version 17. A value of $p \leq 0.05$ was used to determine significance, except for determination of abiotic variables included in the regression models, where $p \leq 0.01$ was used.

RESULTS

Remote sensing data

Results from the MODIS-LAI data demonstrate that LAI was consistently higher at Pink Valley (Figure 2) than at Allanwater (Figure 3) for the available time period of 2000-2010. The mean LAI value was also significantly higher overall at Pink Valley than at Allanwater (t = 3.80, p <0.000, n = 441). Moreover, the data at both sites showed a trend of decreasing LAI during this time period, which was significant (p = 0.03) at Pink Valley.

INSERT FIGURES 2 AND 3 HERE

Data on shrub abundance from analysis of aerial photographs are presented in Table 1.

INSERT TABLE 1 HERE

Shrub counts showed an overall decrease during the period 1968-2004 at Pink Valley, whereas there was a substantial increase at Allanwater. In both cases these changes were most pronounced in the period 1968-85, although at Allanwater the 1968-1975 period was characterised by almost no change in shrub numbers, whereas between 1975 and 1985 numbers increased markedly. From 1985-2004 both sites demonstrated relatively little change in shrub abundance.

Vegetation transect data

Comparison of the two sites using both Sorensen's Index and Renkonen's Index produced values of 0.67 and 64.53% respectively, suggesting strong similarity between the sites in the range and relative abundance of the grass species present. Although there was a greater number of grass species identified at Pink Valley both sites were dominated by perennial species with few annual species recorded (Table 2).

INSERT TABLE 2 HERE

Proportions of rock and bare ground did not differ significantly between sites but overall range condition score was significantly ($p<0.05$) higher at Allanwater (Table 3).

INSERT TABLE 3 HERE

For biotic variables, basal vegetation data is summarised in Table 4. Grasses were significantly more abundant at Pink Valley, whether classified by frequency or cover. They also had a significantly lower overall PTD. Dwarf shrubs, whilst significantly more frequent at Allanwater, showed no difference between sites in terms of cover or PTD. When amalgamated across all life forms, basal vegetation showed significantly greater basal cover and significantly lower PTD at Pink Valley compared to Allanwater. Analysis of abundance data for key species showed that whilst the preferred grazing species *T.triandra* was significantly more frequent at Allanwater than Pink Valley, its overall cover did not differ significantly between the sites and mean PTD was actually significantly lower at Pink Valley. For *E.muticus*, there was no significant difference in frequency, cover or PTD between sites.

INSERT TABLE 4 HERE

The frequency of occurrence of woody shrubs at Allanwater was significantly greater than at Allanwater, although there was no difference in mean distance to shrub stems (Table 5).

INSERT TABLE 5 HERE

Current patterns of rangeland degradation

Multiple linear regression analysis of key degradation indicators at each site revealed which biophysical variables explained most of the current variation in rangeland quality at each site (Table 6).

INSERT TABLE 6 HERE

At Allanwater several biophysical variables appear to explain current patterns of degradation measured at the sample sites. Aspect appears critical, with sites with more northerly aspects having higher levels of dwarf shrub occurrence, larger

PTDs and decreased levels of cover. Elevation is also important, with sites at lower elevations being associated with greater amounts of shrub occurrence and smaller PTDs and higher elevations with increased veld condition scores (VCS). Distance is important in explaining both VCS and, to a lesser extent, dwarf shrub occurrence, with sites with higher VCS located at greater distances from the settlement, and greater occurrence of dwarf shrubs at sites nearer the settlement. At Pink Valley, distance from the homestead seems to be the single most important variable explaining current measures of degradation. More distant sites are associated with greater VCS scores and greater levels of plant cover, whilst sites closer to the homestead had greater levels of dwarf shrub occurrence, although in this case distance was co-linear with elevation. Aspect also explained occurrence of dwarf shrubs, with greater amounts on sites with more northerly aspects. Slope was also important in explaining VCS at Pink Valley, with sites on steeper slopes having lower VCS scores.

DISCUSSION

Landscape scale change

The MODIS data showed a slightly larger but significantly higher overall LAI value at Pink Valley compared to Allanwater, which is consistent not only with greater rainfall at Pink Valley, but also with the greater cover of basal sward at the site and lower occurrence of shrubs. Interestingly, there was also an overall decline in LAI values at both sites during 2000-2010. Whilst this fits well with the observed increase in shrub encroachment at Allanwater, the significant decline in LAI at Pink Valley is more difficult to explain given that shrub density at the site remained relatively stable during this period. However, given the greater proportion of grassland at the site, compared to shrubs, it is possible that rainfall is having more influence on LAI at Pink Valley. The overall downward trend may therefore be a result of the very high rainfall (> 700mm/annum) recorded at Pink Valley during three of the first five years of this period (2001, 2001 and 2004) and average or below average rainfall during the subsequent period (2005-10). The strong relationship between LAI and rainfall at Pink Valley is corroborated by a significant regression relationship ($R^2 = 0.62$, $p = 0.012$) for the period 2000-2008. It must also be accepted that it is difficult to establish a definitive direction of change at either site with only 10 years of data.

Much more effective in this respect was historical analysis of the aerial photographs at each site. This demonstrated an increase in bush encroachment at Allanwater compared to Pink Valley, as hypothesised. However, although there was a substantial overall increase in bush abundance at Allanwater during 1968-2004, this was not continuous. Bush numbers were quite stable during the period 1968-1975 but increased dramatically during 1975-85, coinciding with the change to communal management at the site, in 1976. The key factor in this may not necessarily have been increased livestock densities (relatively few families settled at the village initially), but rather the removal of fire as a management tool. Burning would have been an integral part of the commercial management system up to 1976 and its absence, combined with a lack of browsing due to a paucity of stock in the early years of settlement, may have encouraged the proliferation of woody shrubs. This is supported by Roques *et al*., (2001), who found that frequent fire was the critical factor inhibiting shrub encroachment over the long term in Swaziland. The relative lack of shrub increase at Allanwater in the years since 1985 may reflect either an increase in browsing pressure, or simply that shrubs had already reached a sufficiently high density at this stage for competition to have become a limiting factor. In comparison, the overall decline in bush abundance observed at Pink Valley, particularly during 1968-1985, was unexpected although this may also be related to the regular burning that was practised historically before the current owner took residence in 1985.

Comparative extent of vegetation degradation and pattern

In comparing the current extent of vegetation degradation, there are clear differences between the sites with regard to indicators based on species composition (quality) and those such as plant cover, PTD and shrub abundance, which are indicative of more fundamental changes in vegetation structure. The almost complete dominance of perennial grasses at both sites is not unexpected and reflects their underlying ecology. Both are essentially stable, equilibrial grassveld sites, receiving relatively reliable annual rainfall. The local veld types are mixed to sour in nature and thus relatively grazing-resistant and predisposed to management techniques such as rotational resting and grazing (Tainton, 1999).

The significantly higher veld condition score at Allanwater compared to Pink Valley is more unexpected. Such high scores are unusual in communal grazing systems in the region. For example, an assessment of the VCS of the nearby Zweledinga site by Vetter *et al*., (1999) gave a mean overall condition score of just 58.15%. It appears that the main driver of the high condition score at Allanwater is the significantly higher relative abundance of *T. triandra*, compared to Pink Valley. As the most important forage species for livestock in these grassveld systems, *T.triandra* scores proportionally very highly in the VCS method adopted in the study. Why *T.triandra* should be so much more abundant at Allanwater, is less apparent. Furthermore, it is clear from the field sampling that high quality patches of forage, typically dominated by *T.triandra*, are not evenly distributed but rather are clumped at the extreme western area of the rangeland at relatively high elevation. The two sites sampled here produced by far the highest condition scores (162% and 142%). This gradient in the distribution of high VCS sites at Allanwater is also supported by the results of the multiple regression analysis at the village. Current grazing management practices may play an important role in the maintenance of such high quality patches of *T.triandra*. The lack of permanent water in these areas means that they are not grazed during the dry period (approx May-September) and their steep topography, combined with distance from the settlement, may prevent intensive grazing even during the summer months. Thus, the fact that the system does not make optimal use of the available forage, as a commercial system might, may enable such key resource patches to be sustained.

The relatively high quality of the grazing resource at Allanwater is also reflected in animal productivity at the village. Figures from the National Wool Growers for the wool clip at Allanwater show a total yield of 10,116kg in 2007 (ECDA, unpublished data). Assuming a similar number of sheep at present to numbers recorded in 2002, this gives a yield per animal of 4.3kg, which is comparable to commercial systems in the region (ECDA, unpublished data).

Despite the current productivity of the rangeland at Allanwater, there are clear signs of vegetation degradation relative to Pink Valley. Grass cover is much higher at Pink Valley and total basal vegetation significantly ($p>0.05$) higher. Although this may, to some extent, be a result of the greater rockiness at Allanwater, the fact that the frequency of grass occurrence was significantly greater overall at Pink Valley suggests that there is an inherent difference in the basal vegetation at the two sites. A key factor appears to be the point to tuft distance (PTD). PTD of both total basal vegetation and grasses at Pink Valley was significantly below that of Allanwater. Furthermore, despite the significantly higher frequency of *T.triandra* at Allanwater, the difference in overall cover at each site was relatively small as a result of the significantly lower PTD of the species at Allanwater. Importantly, for all of these life form and species groups, mean PTD values at Allanwater were above 3 cm, which suggests that rangeland at the site is more vulnerable to erosion than that at Pink Valley (Beckerling *et al*., 1995).

Thus, there is clear evidence of degradation in the basal vegetation layer at Allanwater relative to Pink Valley. Whether this is related to the relatively high grazing pressure at the village is more debateable. However, the patterns identified within both the PTD and cover measurements suggest that grazing pressure is playing a role. The fact that more northerly aspects were associated with higher PTDs and lower cover values (particularly at lower elevations) supports the idea of grazing driven change, as livestock will selectively graze on North-facing slopes, where forage tends to be more palatable (Evans *et al*., 1997). There is also strong support in the literature for degradation of basal vegetation in communal grazing areas relative to commercial farms. Vetter *et al*., (2006), demonstrated significantly increased PTDs for basal vegetation in communal systems in Herschel compared to neighbouring commercial systems. The strongly negative relationship between shrub occurrence and cover at Allanwater also suggests that the significantly increased levels of shrub encroachment documented since communal grazing, may be a result of reduced basal vegetation cover at the site, or vice-versa. Certainly, it is widely recognised that reduced basal vegetation cover favours shrub development both by increasing soil moisture availability and diminishing the frequency and intensity of fires that can control growth of mature shrubs (Scholes and Archer, 1997, Roques *et al*., 2001). Nevertheless, despite the higher overall basal cover at Pink Valley, the analysis shows a pattern of decreased basal cover nearer the homestead. This suggests that in spite of a system of rotational grazing being in place, the camps nearer the homestead continue to receive more grazing even in this commercial system. This probably reflects their increased use during the dry season, particularly for raising winter lambs.

Other key indicators of comparative vegetation degradation at Allanwater are the significantly higher occurrence of both dwarf karroid and woody shrubs relative to Pink Valley, as initially hypothesised. The dwarf shrubs are typically species such as *Chrysocoma ciliata* and *Felicia filifolia*, which are recognised as indicators of overgrazing (Hoffman and Cowling, 1989, Evans *et al*., 1997). The fact that the occurrence of these species was strongly associated with northern aspects and sites nearer to the village, which are most likely to experience the heaviest grazing pressure, further suggests that their occurrence at Allanwater is a result of sustained heavy grazing pressure. Vetter *et al*., (2006) in Herschel and Anderson and Hoffman (2007) in Namaqualand, also found significantly higher abundance of dwarf shrubs at communal sites compared to commercial livestock farms. Although, much diminished in frequency, a similar pattern of distribution of these same dwarf shrub species is evident at Pink Valley, suggesting that injudicious grazing management is also occurring here. The complete absence of fire since the late 1980s may be a key factor in this.

The vegetation degradation suggested by the abundance of woody shrubs at Allanwater is not just quantitative, as the type of species occurring is also critical. At Pink Valley the woody shrubs are represented by a single dominant genus, *Passerina*, whereas at Allanwater, in addition to *Passerina* spp., *Euryops floribundus* is also present at an overall frequency of 6.5% (13.6% of the total shrub occurrence). The presence of *E. floribundus* at is of particular concern, as it is widely recognised to reduce the grazing potential of invaded areas by suppressing grass growth both through shading and the resin-rich leaf litter it produces and, once established, it is very difficult to remove (Vetter *et al*., 1999, Shackelton and Gambiza, 2008).

CONCLUSIONS AND POLICY IMPLICATIONS

The overall findings from the study present a rather mixed picture. Superficially, the situation at Allanwater is one of resilience in productivity evidenced by range condition scores and livestock yields comparable with commercial systems. However, a deeper interrogation of the ecological evidence suggests that 30 years of communal grazing, combined with the absence of controlled burning, have resulted in a substantial loss of ecological integrity in the system. In the early years of communal occupation rapid increases in woody shrubs were apparent as the most obvious signs of degradation but, whilst the pace of shrub encroachment appears to have diminished, declines in key measures of basal vegetation integrity are now evident. Moreover, these show typical signs of being driven by overgrazing, being most prevalent nearer to residential areas, at lower elevations and on north-facing slopes; all areas where livestock will preferentially graze. Thus, although livestock productivity has been maintained in the short term, it must be questioned whether this can be sustained in the medium to long term, given the existing trajectory of degradation of the resource base.

How representative is this study of the broader picture in Eastern Cape and beyond? Certainly, an analysis of just two sites, however detailed, can provide nothing more than an initial insight into the complexity of rangeland change associated with tenure shift. These results need both corroboration and further development through the analysis of 'new' communal sites in different ecotopes, with higher grazing pressures and more *laissez-faire* communal management systems. Allanwater through its inheritance of a high quality grazing resource, and attempts at perpetuating a grazing management system within defined community boundaries, is somewhat anomalous in communal terms. Nevertheless, the level of degradation apparent at Allanwater, even when compared to a relatively poorly-managed commercial system, suggests that there are lessons to be learnt. With current land reform policy focused on accelerating the rate of land transfer to emergent farmers, potential land transfers should be evaluated not just in terms short term economic output and associated livelihood benefits but also, over the longer term, with regard to the ability of beneficiaries to manage the grazing resource such that ecological integrity is maintained. There is therefore an urgent need for a programme of long term range monitoring associated with land transfer, both historical and post-apartheid, in and around 'new' communal areas. Only on this basis can a realistic decision be reached as to whether communal management can deliver both in terms of productivity as well as the effective maintenance of the resource base and provision of associated ecosystem services.

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FIGURES

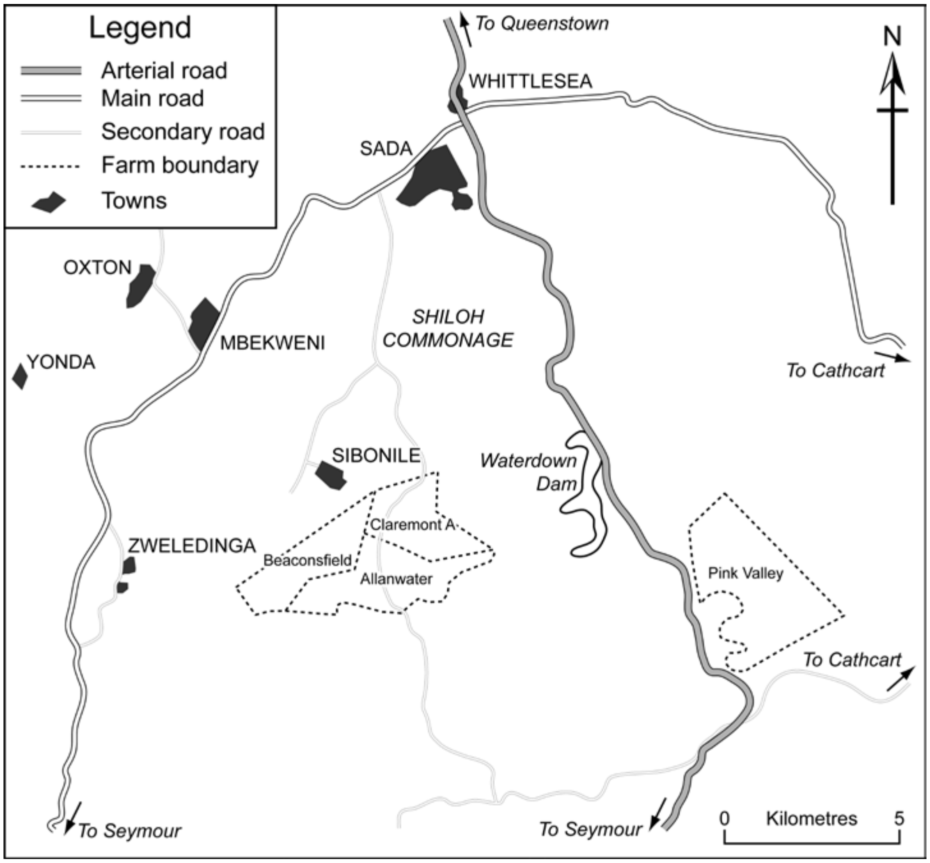

Figure 1: Location of study sites

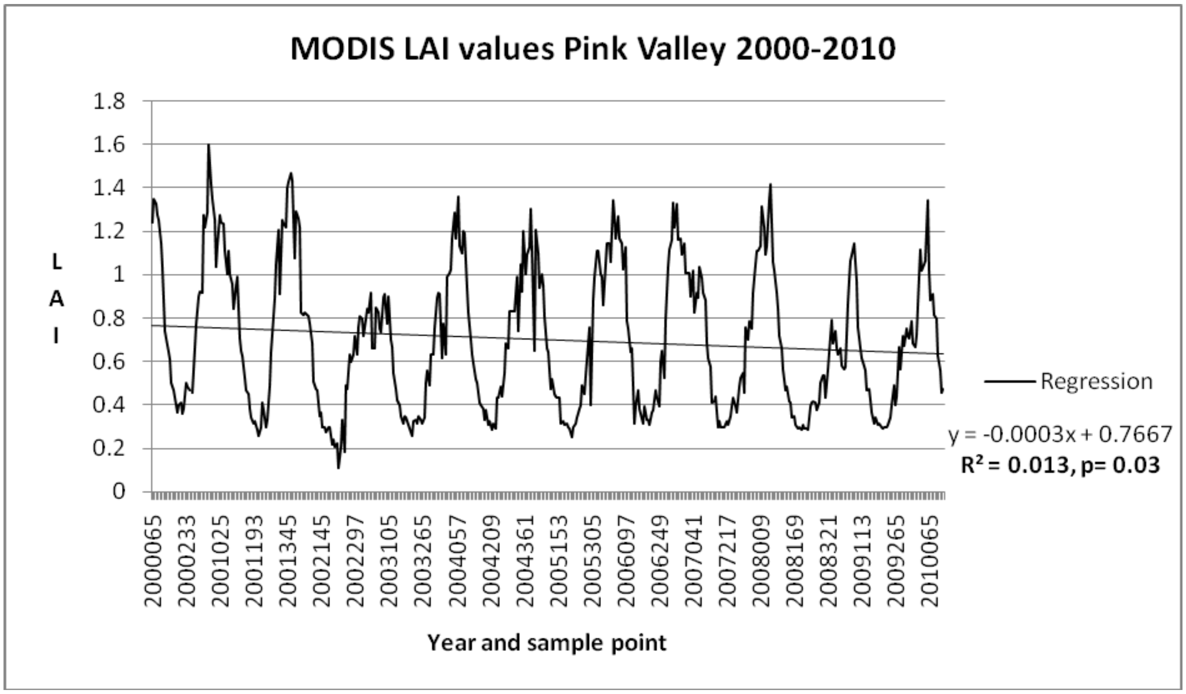

Figure 2: MODIS-LAI data for Pink Valley 2000-2010

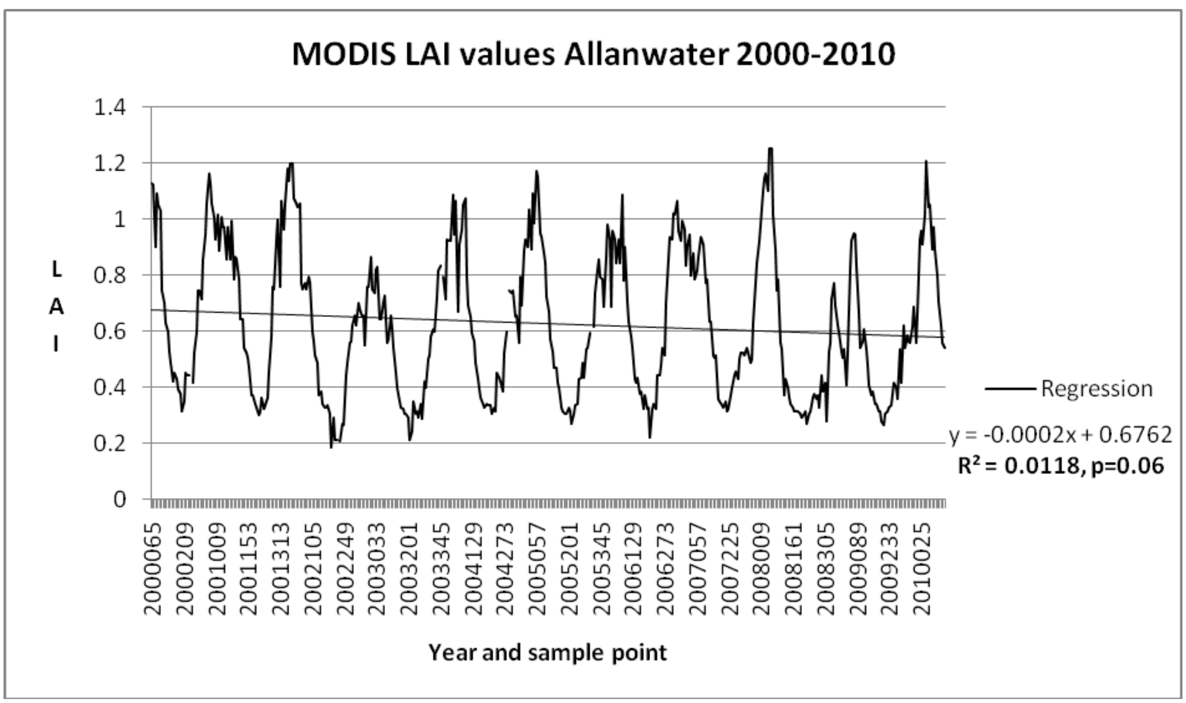

Figure 3: MODIS-LAI data for Allanwater 2000-2010

TABLES

Table 1: Changes in shrub abundance at Pink Valley and Allanwater (1968-2004).

	Total shrub counts				% change				
Site	1968	1975	1985	2004	1968-1975	1968-1985	1975-1985	1985-2004	Overall
Pink Valley (n = 35)	1079	NA	970	941	NA	-10.1	NA	-3.0	-12.8
Allanwater (n = 56)	1509	1505	1828	1867	-0.3	+21.1	+21.5	+2.1	+23.7

Table 2: Numbers of grass species and proportion of annuals and perennials at Pink Valley and Allanwater.

Parameter	Pink Valley	Allanwater
No. grass species	23	16
Perennial species: count (proportion)	23 (100%)	15 (99.7%)
Annual species: count (proportion)	0 (0%)	1 (0.3%)

Table 3: Comparison of abiotic variables at Pink Valley and Allanwater.

Parameter	Allanwater (n=15)	Pink Valley (n=16)	t-value	*p*
Rock (%)	13.67 (±3.25)	8.44 (±2.46)	1.29	0.21
Bare Ground (%)	65.87 (±3.028)	60.38 (±3.62)	1.15	0.26
Mean range condition score (%)	77.70 (±8.59)	53.91 (±6.34)	2.25	0.03

Table 4: Relative abundance measures for basal vegetation by life form and key species at Allanwater and Pink Valley.

Frequency				
Life form/species	Allanwater (n=15)	Pink Valley (n=16)	Z score	*p*
Grasses	89.8 (±1.06)	93.19 (±1.73)	2.32	0.02
Dwarf shrubs	6.67 (±1.11)	3.31 (±1.08)	2.29	0.02
Herbs	2.00 (±0.41)	3.50 (±1.19)	0.49	0.63
Themeda triandra	32.53 (±5.93)	13.13 (±4.30)	2.81	0.01
Elionurus muticus	18.33 (±4.35)	16.38 (±4.16)	0.55	0.58
Cover (%)				
Life form/species	Allanwater (n=15)	Pink Valley (n=16)	Z score	*p*
Grasses	19.33 (±1.63)	29.19 (±3.83)	1.88	0.06
Dwarf shrubs	0.73 (±0.21)	0.69 (±0.38)	1.24	0.21
Herbs	0.40 (±0.16)	1.31 (±0.62)	1.20	0.23
Total basal vegetation	20.47 (±1.73)	31.19 (±3.95)	1.94	0.05
Themeda triandra	7.00 (±1.57)	5.00 (±2.35)	1.83	0.07
Elionurus muticus	3.40 (±0.83)	2.81 (±0.71)	0.36	0.72
PTD (cm)				
Life form/species	Allanwater	Pink Valley	Z score	*p*
Grasses	3.54 (±0.12) (n=1347)	2.94 (±0.10) (n=1486)	4.51	0.00
Dwarf shrubs	7.48 (±0.89) (n=100)	4.43 (±0.67) (n=53)	1.81	0.07
Herbs	2.87 (±0.71) (n=30)	1.63 (±0.24) (n=56)	1.38	0.17
All basal vegetation	3.79 (±0.13) (n=1477)	2.95 (±0.09) (n=1595)	5.45	0.00
Themeda triandra	3.34 (±0.18) (n=488)	2.14 (±0.18) (n=210)	4.15	0.00
Elionurus muticus	3.65 (±0.21) (n=275)	4.18 (±0.25) (n=262)	1.42	0.16

Table 5: Relative abundance of woody shrubs at Allanwater and Pink Valley.

Parameter	Allanwater	Pink Valley	Z score	*p*
Frequency	47.13 (±10.30) (n=15)	14.56 (±7.29) (n=16)	2.52	0.01
Distance	44.90 (±0.99) (n=707)	46.68 (±1.85) (n=233)	0.88	0.38

Table 6: Regression models explaining variation in degradation indicators at both sites (only variables significant at $p<0.1$ are presented).

Site	Indicator	R	F	p	Variable	*t*	p
Allanwater	VCS	0.45	4.92	0.02	Distance	3.79	0.030
					Elevation	1.94	0.079
	Karroid	0.69	16.24	0.00	Aspect	5.69	0.000
					Distance	-1.81	0.095
	Cover	0.65	14.03	0.00	Shrubs	-5.06	0.000
					Aspect	-2.54	0.026
	PTD	0.58	10.65	0.00	Aspect	3.35	0.006
					Elevation	-3.20	0.008
	Shrubs	0.63	24.37	0.00	Elevation	-4.94	0.000
Pink Valley	VCS	0.69	17.63	0.00	Distance	5.78	0.000
					Slope	-3.40	0.005
	Cover	0.51	16.28	0.01	Distance	4.04	0.010
	Karroid	0.43	4.78	0.02	Distance*	-3.27	0.007
					Elevation*	2.55	0.025
					Aspect	1.93	0.077

*Co-linearity in variables