Soil quality gradients around water-points under different management systems in a semi-arid savanna, South Africa

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Abstract

Over 70% of South Africa is too arid for crop farming and is used for either commercial livestock ranching, communal livestock ranching, or game ranching. The inherently different management characteristics of these ranching systems and their effects on vegetation dynamics makes rangeland degradation a contentious issue. We used 500-m-long grazing gradients around water-points to evaluate the effects of management type on soil quality. Results showed significant negative effects of management type on soil parameters (i.e. soil pH, nitrogen, and organic carbon) within 0–100 m from the water-point. Commercial livestock ranching had the greatest negative effect on the immediate area around the water-point. Beyond 100 m, no effect of herbivore activity on soil parameters was detected under any management system.

Keywords: Semi-arid rangeland; Pastoralism; Wildlife; Rangeland degradation; Soil condition; Grazing gradients

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1. Introduction

Rangelands are important ecosystems for South Africa. More than 75% of South Africa’s land surface is used for pastoral production (Tainton, 1999). This is mainly due to the low precipitation in these areas, which makes the land unsuitable for crop farming. Three main rangeland management systems can be found in South Africa: commercial livestock ranching, communal livestock ranching, and game ranching. These three systems differ in management structure, animal diversity, management of grazing resources, and products (Table 1).

Of the three livestock management systems, communal livestock ranching has most often been criticized for its perceived degrading effects on rangelands (Ellis and Swift, 1988; Behnke and Scoones, 1992). In the ‘tragedy of the commons’ (Hardin, 1968), it is argued that rangeland degradation of communal lands is nearly inevitable due to the many individuals that utilize the same land. The theory reasons that it is beneficial for an individual livestock owner to be selfish and overstock the ‘commons’ (i.e. communal lands) rather than to apply conservative stocking rates because the benefits thereof accrue to the individual but the costs of heavy stocking are shared by all. This then leads to overstocking and eventually to irreversible degradation (Hardin, 1968). African communal rangelands frequently support high numbers of livestock, and often exceed advised carrying capacity levels (Abel, 1993; Scoones, 1993; Tapson, 1993; Ward et al., 1998). Because of the high stocking rates on communal lands it has often been assumed that communal livestock management systems cause rangeland degradation (Ellis and Swift, 1988; Behnke and Scoones, 1992; Ward et al., 1998; Dougill et al., 1999). Destocking programs have been initiated to prevent rangeland degradation but many have failed to reach their objectives (Abel, 1993). Contrastingly, commercial cattle ranchers apply much lower stocking rates in order to produce high-quality products for markets.

1.1. Rangeland degradation

Traditionally, arid and semi-arid rangeland vegetation dynamics have been explained using equilibrium models (Ellis and Swift, 1988; Behnke and Scoones, 1992). However, recent studies have shown that rangeland degradation is caused by overgrazing, which leads to a decline in vegetation cover and productivity. This process is accelerated by climate change, which is causing increased temperatures and decreased precipitation. As a result, rangelands are becoming more susceptible to degradation, which threatens the livelihoods of the people who rely on them for grazing and other uses. To address this issue, it is important to develop sustainable grazing management practices that take into account the complex interactions between climate, vegetation, and human activities.
1992). Equilibrium models reason that vegetation dynamics are regulated by biotic controls such as herbivory and that vegetation tends towards an equilibrium (Clements, 1916). Vegetation moves in a predictable manner along a series of successional stages, beginning at the pioneer stage and ending at the climax stage. Change from one stage to another is driven by biotic events (Clements, 1916). It has long been assumed that grazing is an important modifier of rangeland vegetation, and therefore also a root cause of rangeland degradation where heavy stocking occurs (Dyksterhuis, 1949; Ellison, 1960; Stoddart, 1960; Holechek et al., 1995). Management systems have long been based on this assumption, e.g. carrying capacity (Stoddart, 1960) and rangeland condition (Dyksterhuis, 1949) concepts have been important management tools to regulate grazing intensity and monitor grazing resources to prevent degradation (Joyce, 1993; Holechek et al., 1995; Tainton et al., 1999).

Ellis and Swift (1988) reviewed the issue of stability of African pastoral ecosystems, based largely on their long-term study in the Turkana region of Kenya. They questioned the widespread assumption that communal pastoralism has degrading effects on ecosystems. They argued it did not because of the highly variable nature of vegetation dynamics of arid and semi-arid rangelands occupied by most African communal pastoralists. Ellis and Swift (1988) argued that abiotic events had a much more important role to play in the vegetation dynamics of these systems than did grazing. It is well known that rainfall in arid and semi-arid ecosystems is highly variable in time and space (Noy-Meir, 1973). Ellis and Swift (1988) reasoned that these climatic attributes are the driving force behind vegetation dynamics; and not biotic events, such as grazing (Ellis and Swift, 1988; Behnke and Scoones, 1992). The stochastic nature of the climatic factors results in erratic vegetation changes and a situation where the vegetation is unlikely to reach an equilibrium state (Ellis and Swift, 1988; Westoby et al., 1989; Behnke and Scoones, 1992; Milton and Hoffman, 1994). Therefore, these ecosystems are called non-equilibrium systems (Ellis and Swift, 1988; Behnke and Scoones, 1992). Changes in non-equilibrium vegetation have been described by the state-and-transition models (Westoby et al., 1989; Milton and Hoffman, 1994). These models allow certain events to cause transitions from one vegetation state to another, and once a transition has occurred, vegetation states may persist for a long period and may even be irreversible (Westoby et al., 1989).

Non-equilibrium systems are thought to have weak or no density-dependent processes between vegetation and herbivore populations (Ellis and Swift, 1988; Illius and O’Connor, 1999:809). Traditional rangeland management tools, such as carrying capacity and the rangeland condition concept, are all based on density-dependence processes (Dyksterhuis, 1949; Jones and Sandland, 1974). These management tools would therefore be of little use in non-equilibrium rangelands. The non-equilibrium concept caused many rangeland scientists to re-evaluate the effect of stocking rates on arid and semi-arid rangelands (i.e. Behnke and Scoones, 1992; Abel, 1993; De Leeuw and Tothill, 1993; Illius and O’Connor, 1999). For example, Tapson (1993) found that herd size on KwaZulu’s communal lands remained stable over the period 1974–1988 contra the conventional expectation that
sustained heavy stocking had caused a decline in vegetation productivity, resulting in decreased animal production. Similarly, Ward et al. (1998) did not find a difference in degradation when comparing commercial ranches and a large (117 000 ha) communal ranch, differing three- to ten-fold in stocking densities. This suggested that stocking densities did not influence rangeland degradation.

An exact border between equilibrium systems and non-equilibrium systems cannot easily be determined. Wiens (1984) suggests that rangelands are located somewhere on a continuum between equilibrium and non-equilibrium dynamics. Therefore, it is not always clear to what extent a system is dominated by either equilibrium or by non-equilibrium forces; and how successful traditional management tools are.

Illius and O’Connor (1999) discussed the relevance of non-equilibrium models for semi-arid rangelands. They concurred with Ellis and Swift (1988) and Behnke and Scoones (1992) that there is a need to review the applicability of equilibrium models to arid and semi-arid rangelands. However, they also believe that density-dependent processes are more important than Ellis and Swift (1988) and Behnke and Scoones (1992) portray. In their article, Illius and O’Connor (1999) argue that density-dependence can occur in arid and semi-arid rangelands. Arid and semi-arid rangeland vegetation is spatially heterogeneous. Vegetation associated with drainage lines, pans, and evergreen trees can function as key resources for herbivore populations (Illius and O’Connor, 1999). The availability of such resources determines herbivore population size (Illius and O’Connor, 1999). There is a density-dependent relationship between herbivore population and key resources. During long droughts, this relationship becomes more apparent (Illius and O’Connor, 1999). They therefore believe that: “... spatially and temporally, the whole system is heterogeneous in the strengths of the forces tending to equilibrium, these diminishing with distance to watering and key-resource areas” (Illius and O’Connor, 1999, p. 809). In other words, vegetation dynamics in key resource areas exhibit equilibrium dynamics, but outside these key resource areas exhibit non-equilibrium dynamics. Furthermore, they argue that “rainfall variability has profound effects on annual variation in species abundance, but, unless there is a directional trend in rainfall, and despite the potential for large annual changes, there is no net change in species composition in the long term” (Illius and O’Connor, 1999, p. 803). Grazing, on the other hand, may have only a small annual effect on species composition. However, the effects of grazing on a species do not change (due, for example, to forage preferences) and, therefore, grazing has a cumulative effect on species composition that can become substantial over long periods of intensive grazing (Illius and O’Connor, 1999).

1.2. Grazing gradients

The present study aims to compare rangeland degradation between three—management systems. Due to the uncertainty regarding the extent that semi-arid rangelands are regulated by equilibrium or non-equilibrium dynamics, rangeland degradation is a contentious issue. Grazing gradients can be used to correlate degradation to a particular management system (i.e. with a specific grazing intensity
and herbivore species) in arid and semi-arid environments where vegetation and soil quality are spatially variable. Gradients starting at water-points show higher grazing intensity and rangeland degradation near to the water-points compared to further away from (Tolsma et al., 1987). These grazing gradients are also called ‘piospheres’ (Lange, 1969). Many studies have used grazing gradients to evaluate the effect of grazing on vegetation (Walker and Heitschmidt, 1986; Tolsma et al., 1987; Andrew, 1988; Jeltsch et al., 1997; Moleele and Perkins, 1998; Thrash, 1998; James et al., 1999; Thrash, 2000; Riginos and Hoffman, 2003). As discussed above, vegetation is profoundly affected by variations in rainfall (Ellis and Swift, 1988; Behnke and Scoones, 1992; Illius and O’Connor, 1999). Recording vegetation composition is, therefore, a snapshot of a short-term situation and does not necessarily reflect the long-term situation. In this study, we used soil indicators to determine rangeland health. Soil indicators are reliable indicators of long-term soil degradation because they are not affected by short-term rainfall fluctuations and drought to the degree that vegetation composition is (Dougill et al., 1999; Turner, 1999).

2. Methods and materials

2.1. Study site

The study site is located 30 km north-west of Kimberley (Northern Cape, South Africa) and focuses on Pniel Estates near Barkly West, (Fig. 1). The coordinates of the centre of the study site are: 28°36’ S, 24°28’ E (1125 m a.s.l.). The climate can be characterized as semi-arid with a mean annual precipitation of 388 mm (C.V. 39%) (Kraaij, 2002) that mainly falls between November and April. Soils are mainly clayey along the banks of the Vaal River and near andesite outcrops. Further away from the river, soils are mainly deep red Kalahari sands with some calcrete pans (shallow, seasonally inundated water bodies). The vegetation is classified as Kalahari thornveld (Acocks, 1988) or Kimberley thorn bushveld (Low and Rebelo, 1998) which includes Acacia erioloba and Acacia tortilis savanna, and Acacia mellifera shrubland. Dominant grasses are Aristida congesta, Eradrostis lehmanniana, and Schmidtia pappophoroides. Bare soil frequency (number of bare soil counts per 30 observations) was significantly higher within 100 m of water-points. Also communal livestock ranching showed higher frequencies of bare soil along the entire transect, compared to the other two management types (Smet and Ward, 2005). Pniel Estates is about 25 000 ha in extent and is comprised of:

1. 3000 ha communally managed livestock ranch; 320 Large Stock Units (LSU) (9.375 ha LSU⁻¹).
2. 10 000 ha commercial game ranch; 912 LSU (10.965 ha LSU⁻¹).
3. 12 000 ha commercial cattle ranch; 750 LSU (16 ha LSU⁻¹).

The Evangelical Lutheran Church of South Africa owns the estate, including the communal area. Five properties border the Estate; viz. Vaalbos National Park,
Rooipoort game ranch, Platfontein communal game ranch (owned by the !Xu and Khwe San (Bushman) communities), and two privately owned cattle ranches (Secretaris and Nooitgedacht). North of Pniel Estates lies the Vaal River (Fig. 1).

For each management system, three water-points were chosen, distributed over the particular ranches and all on clayey soils. For each management system, an additional water-point was sampled outside Pniel Estates, for the communal ranch this was Camp 11 of the Barkly West commons (coordinates: 28°29'27.56" S, 24°30'02.50" E), for the commercial ranch we chose Nooitgedacht (coordinates: 28°38'21.08" S, 24°36'18.75" E), and for the game ranch the Rooipoort game ranch (coordinates: 28°36'51.89" S, 24°21'54.54" E) was used (Fig. 1). In total, nine water-points were sampled on Pniel Estates and three on other ranches. At each water-point, three 500 m transects were set out, radiating from the water-point. Soil samples were collected at fixed distances from the water-point (i.e. 0, 25, 50, 75, 100, 150, 200, and every 50 m thereafter to 500 m). At every distance, three soil samples were collected to a depth of 10 cm. The first sample was collected on the transect and...
the others 5 m left and right of the transect. The three samples were combined into one composite sample. Soils were dried (40 °C for 24 h) and kept dry at room temperature before being analysed. Next to the parameters discussed in this article, vegetation parameters were collected along the same transects at the same time. The results of the vegetation study are discussed in Smet and Ward (2005).

2.2. Soil analysis

Of the 12 water-points, only a sub-sample of soil samples collected along the transects was analysed. In each management system, two water-points on Pniel Estates and one outside Pniel Estates was chosen. At each of these water-points, one transect was selected at random, bringing the total number of transects to three per management system. The soil samples of these transects were analysed by a commercial laboratory. The following soil nutrients and characteristics were analysed: soil pH$_{KCl}$, resistance, total nitrogen (N), available phosphorus (P), and organic carbon (C). Soil resistance (Ω), the inverse of soil electrical conductivity (Siemens (S) = Ω$^{-1}$), is a measure of the concentration of ions present in the soil and their electrical charge (Rowell, 1994), which is often used to measure salt content of soils and an indication of desertification. Total soil N was determined using Dumas’ total combustion method (Dumas, 1831; Bremner, 1996) with a Leco nitrogen analyser. Available P was analysed using the Bray II method (Bray and Kurtz, 1945). Organic C was determined using the Walkley–Black method (Walkley, 1947; Nelson and Sommers, 1996).

2.3. Soil bioassay

Each composite sample was transferred into a black plastic bag (dimensions: 19 cm high x 8 cm wide x 5 cm deep) after which the bags were tagged for identification. These were then randomly placed on a nursery bed. The grass species *Eragrostis tef* was sown in each of the bags. This species was chosen as it is closely related to species commonly found in our study (i.e. *E. lehmanniana*). *E. tef* is, therefore, likely to respond in similar ways as the natural occurring species do. Furthermore, the species is commercially available, and commercial seeds are more likely to be similar to each other than seeds collected from the rangeland. This reduces the risk that bioassay results are biased because of variance in seed quality. Bags were watered twice daily using an automatic sprinkler system. Grasses were left to grow for 2 months before being harvested. Plants were washed and roots separated from the shoot. Roots and shoot were stored separately in paper bags and dried for 24 h at 70 °C. After drying the roots and shoots, their respective dry masses were recorded.

2.4. Statistical analysis

Variables were tested using analysis of covariance (ANCOVA). In these tests, management type and distance from the water-point were categorical and continuous predictors, respectively. We tested various variables using broken-stick
regression models. The procedure we used was as follows: We determined the approximate breakpoint of the regression by eye and then calculated the error M.S. for the linear regressions for the data points to the left and to the right of this point. We then calculated the pooled error M.S. for these two regression lines. After this, we shifted the breakpoint one sampling unit to the left of the initial breakpoint and recalculated the pooled error M.S. as above. Then we recalculated the pooled error M.S. using a breakpoint one sampling unit to the right of the initial breakpoint. The breakpoint with the lowest pooled error M.S. (i.e. the broken stick regressions explain most of the variance in the data) was taken as the distance at which the water-point effect ended (see Ward and Pinshow (1994) for further explanation of this statistical analysis).

3. Results

3.1. Soil pH

There was a significant effect of management type on soil pH (ANCOVA: \(F = 3.306, p = 0.040, \text{df error} = 109\)). There was also a significant effect of the covariate (ANCOVA distance: \(F = 44.017, p < 0.001\)). Soils around the water-points in the communal livestock ranches had lower soil pH values than the other two management systems. However, there was no significant difference in the effects of commercial cattle and game ranching on soil pH (ANCOVA: \(F = 0.051, p = 0.822, \text{df error} = 71\)).

As can be seen from Fig. 2, soil pH values were high around water-points, followed by a steady decrease in pH up to 100 m from the water-point. After this distance, soil pH reached an asymptote. The broken-stick regression confirmed the breakpoint to be at 100 m from the water-point.

3.2. Soil resistance

Management type did not have a significant effect on soil resistance (ANCOVA: \(F = 2.962, p = 0.056, \text{df error 109}\)). However, there was a significant effect of the covariate on soil resistance (ANCOVA distance: \(F = 15.472, p < 0.001\)). Soil resistance was low near water-points and increased to 100 m from the water-point (Fig. 3). No further change was found beyond 100 m from the water-point.

3.3. Soil nitrogen

Management type had a significant effect on soil N (ANCOVA: \(F = 4.932, p = 0.015, \text{df error} = 109\)), as did the covariate (ANCOVA distance: \(F = 13.874, p < 0.001\)). Here, commercial cattle ranching had the highest soil N concentrations, whereas communal livestock and game ranching had comparable levels. Soil N concentration in the vicinity from the water-points (i.e. 0–75 m) in the commercial cattle ranches was many times higher than in the other two systems.
Fig. 2. Soil pH (KCl) change along transects in commercial cattle ranching (●), communal livestock ranching (■), and game ranching (△) management systems.

Fig. 3. Soil resistance (Ω) change along transects in commercial cattle ranching (●), communal livestock ranching (■), and game ranching (△) management systems.
At 75 m from the water-point, this returned to the same levels as under the other two management types.

**3.4. Soil phosphorus**

Soil phosphorus was not affected by management type (ANCOVA: $F = 2.169$, $p = 0.119$, df error = 109), but was affected by the covariate, distance (ANCOVA distance: $F = 16.773$, $p < 0.001$). Within 75 m of the water-point there is a change in soil P concentrations, except for communal livestock farming where the change is minimal (Fig. 5).

**3.5. Soil organic carbon**

There was a significant effect of management system on soil organic C (ANCOVA: $F = 3.782$, $p = 0.026$, df error = 109), as well as an effect of distance on soil organic C (ANCOVA distance: $F = 19.136$, $p < 0.001$; Fig. 6). Soil organic C was high within 75 m from water-points in the commercial cattle ranches after which soil organic C decreased and stabilized. No effects were observed along the transects of the other two management systems.

**3.6. Bioassay**

Management system did not have a significant effect on shoot dry mass (above-ground mass) (ANCOVA: $F = 2.960$, $p = 0.053$, df error = 356), whereas there was
Fig. 5. Available soil P (mg kg⁻¹) change along transects commercial cattle ranching (●), communal livestock ranching (■), and game ranching (△) management systems.

Fig. 6. Soil organic C (%) change along transects commercial cattle ranching (●), communal livestock ranching (■), and game ranching (△) management systems.
an effect of distance on shoot dry weight (ANCOVA distance: $F = 22.857$, $p < 0.001$). Shoot dry mass followed similar trends to those for soil nutrients. For commercial cattle ranching, we found high values within 75 m of the water-points, after which mass stabilized at lower values. Under the other two management types, shoot mass remained stable along the entire transect.

Management system did not have an effect on root dry weight (below-ground mass) (ANCOVA: $F = 2.522$, $p = 0.082$, df error = 356). There was a significant effect of distance from the water-point (ANCOVA distance: $F = 9.063$, $p = 0.002$, df error = 356—Fig. 7). Dry root mass stabilized at 75 m.

Management system did not significantly affect root:shoot ratio (ANCOVA: $F1.876$, $p = 0.155$, df error = 344). There was no effect of distance from the water-point on root:shoot ratio (ANCOVA distance: $F = 1.370$, $p = 0.242$—Fig. 8). Fig. 8 shows the root:shoot ratio values along transects of the three management types. No pattern in the values of the different management types could be distinguished.

4. Discussion

4.1. Soil quality gradients

Change in soil properties and nutrients generally occurred within 100 m of the water-point. Similar ranges have been recorded by other studies: For example,
Tolsma et al. (1987) found changes in soil nutrients to occur up to a distance of 100 m from the water-point, while Turner (1998) and Dougill et al. (1999) found changes within a distance of 200 m from water-points. Centripetal movement of nutrients towards these zones around water-points has been attributed by Tolsma et al. (1987) and Turner (1998) as the reason for the increase in soil nutrients near water-points. All of the soil properties and nutrients can, in some way or other, be influenced by herbivore activity. Herbivore grazing, trampling, defecation, and urination can affect soil pH (Killham, 1994), resistance (Hao and Chang, 2003), N (Whitehead, 2000), P (Scholes and Walker, 1993), and organic C (Snyman, 1999). These properties therefore gave us a good indication that herbivore activity did affect soil quality. Similar trends were found in the vegetation data (see Smet and Ward, 2005).

4.2. Commercial cattle ranching

In this study, increased nutrient levels around water-points occurred under commercial cattle ranching in particular. There is a possibility that the increased nutrients close to the water-points are partly derived from the outer regions of the grazing area, as suggested by Tolsma et al. (1987) and Turner (1998), through centripetal movement of nutrients. However, a more likely source of nutrients around the water-points is the supplementary feed that is given to cattle at water-points in the commercial cattle ranches. Nitrogen levels around water-points on the commercial cattle ranches were many times higher than those in the communal
livestock and game ranches. These particularly high levels are probably partly caused by the urea supplement given to cattle during the dry period as N source (Meissner, 1999). Similarly, low soil resistance might also be caused by the supplementary feeds. However, high salt content in the ground-water might also be a cause of the low resistance.

High organic carbon near water-points is likely to be caused by centripetal movement. Cattle get most of their roughage from vegetation in the grazing areas and regular visit the water-point. Here they deposit organic matter through their faeces. High organic C percentage increases soil fertility (Stewart et al., 1987; Berg et al., 1997; Snyman, 1999) and affects soil pH (Killham, 1994). Soil organic C can buffer acidity and retain a neutral soil pH level (Bloom, 2000).

The soil bioassay recorded a good growth of $E. tef$ within 100 m of the water-point. This is most likely to be caused by the good soil fertility, e.g. high N, P, organic C, and a high pH. Soils near water-points showed low soil resistance indicating elevated salt concentrations. If these salt effects had negatively affected vegetation growth near water-points we were not able to record this. The absence of negative effects of low soil resistance on grass growth is probably caused by the regular watering of the plants in the bioassay, which may have washed out salts.

### 4.3. Communal livestock ranching

Soils around the water-points in the communal livestock ranches did not have low soil resistance, which indicates low soil salinity. This is different from the situation on both commercial cattle and game ranches, which did show low soil resistance around the water-points. This difference is most probably caused by a number of factors: (1) livestock are not supplemented with salts at the water-points, (2) water-points are rain-fed, whereas the water-points in the other management systems provide borehole water which is slightly brackish, and (3) because water-points are rain-fed, communal livestock can only use the water-points for a couple of months per year. This situation differs from areas where there is no access to natural water sources such as rivers. In those cases pressure on vegetation around water-points may be sustained for longer periods.

When the water-points in the communal ranches run low on water, or completely dry-up, due to drought, livestock rely on water from the bordering Vaal River. Therefore, it is likely that herbivore activity around these water-points is not much higher than in the surrounding areas, except for certain periods when water-points contain water. Thus, distance to water-point has little effect on soil parameters and nutrients. The lower intensity of livestock around the water-points is the most likely reason that soil quality is not affected as much as in the other management systems. On the communal ranches, rangeland degradation might be centred around other key resource areas, such as $A. erioloba$ woodlands on deep sands or riparian vegetation near the Vaal River. We note in this regard that an early resident of Pniel Estates, Solomon Tshekisho Plaatje, recorded that the $A. erioloba$ woodlands were heavily utilized by communally managed livestock more than 100 years ago, indicating their relative resilience to heavy grazing pressure (Willan, 1996). A more
recent study (Britz, 2003) has shown that communally managed *A. erioloba* woodlands still produce a high biomass of palatable grasses in spite of prolonged heavy grazing but do have higher abundances of poisonous and unpalatable species than similar woodlands under game ranching.

4.4. Game ranching

In contrast to commercial cattle ranching, game ranching had only limited effects on soil properties and nutrients. Game also receive some supplementary nutrients, although in much lower quantities. The water-points that were used were mostly visited by water-independent game that do not stay near the water-point for long periods (Grossman et al., 1999). It is also important to note that the water-points were commercial cattle water-points up to 1989.

Soil P concentration around water-points in the game ranches was higher than under communal livestock ranching and lower than under commercial cattle ranching. The explanation for the high soil P concentrations could lie in the fact that these water-points were used for commercial cattle ranching in the past. Soil P does not easily leach in semi-arid environments and P volatilization does not occur (Whitehead, 2000). The high soil P around water-points may, therefore, be caused by the previous management system. Total soil N did not show any effect near the water-point. Soil N is very prone to volatilization and leaching (Scholes and Walker, 1993). Because of this, it is unlikely that high levels of total soil N, accumulated in the soil by commercial cattle ranching in the past, would remain in the soil as long as soil P does. Soil pH around the water-point is still relatively high compared to the communal livestock ranches, which might have to do with a higher percentage of calcrete in the vicinity of the game water-points, and not with increased nutrient levels as might be the case for the commercial cattle ranches. Due to the low availability of nutrients, bioassay results did not differ over the length of the transect, indicating no effect of herbivores.

We may conclude that when commercial cattle ranching is not practiced in the area, soil nutrient levels and soil physical properties will be restored to normal levels. Total soil N and organic C recover in a relative short time period (<10 years) and soil P probably over a longer period (>10 years) due to its low mobility (Whitehead, 2000).

4.5. Rangeland degradation

Our results show that near the water-points there is a clear effect of herbivore activity on soil properties and nutrients, and that management systems seem to have different effects on soil parameters. Effects among management systems and along the transects can also be seen in vegetation parameters (Smet and Ward, 2005). We therefore conclude that rangeland degradation under the present conditions in the study area is influenced by herbivore activity. We concur with Illius and O’Connor (1999) that the availability of key resources in the area allows for density-dependent coupling between the key resources and the herbivore population.
Due to the availability of key resources in the study area, rangeland degradation is not solely or predominantly caused by non-equilibrium dynamics, as proposed by Ellis and Swift (1988). Rather, equilibrium dynamics seem to have an important role to play in rangeland degradation, affecting soils as observed from our results and vegetation as described in Smet and Ward (2005).

4.6. Practical implications

The soil quality gradients around water-points showed that livestock and game had clear effects on soil parameters. Acknowledging that livestock and game affect rangelands has consequences for the way managers manage their rangeland. Unlike Ellis and Swift’s (1988) proposal, stocking rate does affect certain parts of the rangeland. Illius and O’Connor (1999) argue that key resources are areas that are primarily prone to degradation due to high stocking rates. It is therefore important for managers to monitor those areas that can be classified as key resource areas. In this study, *A. erioloba* woodlands and riparian vegetation along the Vaal River may function as key-resource areas. Areas that are not likely to be key-resource areas are probably less prone to degradation due to lower use of these areas in all but the driest periods and the recharging effect of good rains (Ellis and Swift, 1988; Ward et al., 1998).

Future studies need to determine the events driving ecosystem change in rangelands. This is important as there is no simple criterion that determines whether a system is predominantly driven by biotic or abiotic events (Illius and O’Connor, 1999). Soil quality gradients around key resources are able to provide quick and useful clues to answer part of this question. In the upper millimetres of soils in semi-arid and arid rangelands cyanobacteria, algae, microfungi, lichens, and bryophytes may form living soil crusts (Belnap et al., 2001). Many of the organisms that form the soil crusts have the ability to fix atmospheric nitrogen and may therefore be important to ecosystem resilience (Aranibar et al., 2003). The occurrence of soil crusts along grazing gradients can be a good additional indicator of level of disturbance (Aranibar et al., 2003; Dougill and Thomas, 2004). Moreover, where soil crusts occur, the top millimetres of the soil may be an important source of nutrients to plants and therefore important to sample independently. Measurement of additional soil parameters, such as soil hydrological characteristics, might give an even better understanding of the effect of herbivore activity (such as trampling) on soil condition (Dougill et al., 1999).

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