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An assessment of ecological impacts of community-based restoration on communal grasslands in the Drakensberg foothills.

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Submitted in partial fulfilment of the requirements for

MASTERS DEGREE IN CONSERVATION BIOLOGY

In the Department of Zoology

UNIVERSITY OF CAPE TOWN

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April 2011

Declaration:

Plagiarism Declaration

> I know the meaning of plagiarism and declare that all of the work in the dissertation, save for that which is properly acknowledged, is my own.

Signature: ______________________________________

Date: Friday, 01 April 2011

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Acknowledgements

This study was funded and commissioned by Asset Research, through The Water Research Commission (Key Strategic Area: Water Utilisation in Agriculture)

I would like to express my sincere gratitude to the following individuals and organisations for their invaluable contribution during the planning, fieldwork and write-up of this thesis.

The Okhombe Monitoring Group, its committee members, team leaders, and the people whom have for many years carried out extensive restoration efforts in and beyond the ward of Okhombe.

Dr. Terry Everson, for her groundbreaking work in the community of Okhombe, and for her valuable insights and inputs throughout my dissertation. Also, for introducing me to the community at Okhombe, and for the use of extensive field equipment from the UKZN grassland laboratories. I am truly grateful for the opportunity to have worked on such an exciting project with such wonderful people.

Prof. M Timm Hoffman, for his patience, support and assistance in both the planning and write up phase of my dissertation. For his insights and feedback on my manuscripts, and for providing office and administrative resources.

Dr Vincent Chaplot for his extensive support and advice with all matters pertaining to pedology and for being so open to collaboration. Prof. Simon Lorentz for his time, equipment, advice and assistance with all matters pertaining to hydrology. Prof Colin Everson for his input and advice in the early stages of project planning. Prof. Kevin Kirkman for his help in identifying many of my collected grass specimens. Mr Craig Morris for his discussion and input around my field work, and endless high quality coffee supply. Petra de Abreu for her assistance with fieldwork. My friends and family, and my fellow CB2010 students for their encouragement and support. Asset Research and its members, for their extensive support and assistance throughout the project.

And last but by no means least; I would like to express my boundless gratitude and respect to my parents, Bronwyn and Reinhard Marx. Without their influences and support, I would never have grown to love and cherish the natural world around me, and would not have chosen to follow the path which has led me to where I am today.

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Abstract

Okhombe is a ward in the Northern Drakensberg where community based restoration of degraded lands has been conducted for over a decade. In this important water supply region, payment for ecosystem services has been suggested as a means through which to conserve biodiversity, improve veld condition, provide income to local communities, and ensure water security. However, before such alternative market mechanisms can be considered, the effects of community based restoration must be properly quantified. The primary aim of this study was to determine some of the ecological impacts of community based restoration in these communal grasslands. A wide range of indicators were used to assess the function of ecosystem processes and health, including a rangeland health assessment technique - Landscape Functional Analysis. Overall, plots within both open rangeland and restored exclosures within Okhombe were found to be well below the level of ecosystem health and function seen at a reference site in Royal Natal National Park. Average infiltration rates within restored plots were consistently higher than those within open rangeland, indicating that restoration action has enhanced this important ecosystem process. Increased variation in measurements of water infiltration on restored plots may be indicative of the recovery of important soil processes such as bioturbation. Basal cover revealed the dominance of large swards of the exotic grass *Paspalum notatum* on heavily grazed open rangeland, although these did not appear to enhance abiotic function. These lawns are suggested to represent an alternative stable state, and warrant further investigation for application as a restoration technique. Species diversity, veld condition and grazing capacities at all sites (both open rangeland and restoration exclosures) within the communal rangelands was relatively low, and further management of grazing pressures and controlled burn and rest cycles are advised to promote the growth of palatable species. Other variables measured, namely, organic carbon, nitrogen, and bulk density failed to show a clear trend between restored and open rangeland. Landscape Function analysis was shown to be an unreliable tool in the evaluation of restoration success in this study, with no correlation between its qualitative indices and traditional quantitative data. The importance of using both biotic and abiotic factors to evaluate ecosystem processes was highlighted on plots within open rangeland dominated by *P. notatum*, as different indicators suggested different degrees of ecosystem function. The difficulty in using different indicators to demonstrate delivery of ecosystem services was exhibited in the lack of agreement between the wide range of indicators assessed. Lastly, this study established important baseline data which will facilitate future research and monitoring.

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1. Chapter One: General Introduction and Literature Review

1.1. Study objectives and approach

In order to properly assess the ecological impacts of the work that the Okhombe monitoring group (hereafter OMG) are carrying out, a wide range of variables needed to be assessed. A broad, multi-faceted approach was considered most appropriate, where multiple factors critical to restoration success were investigated. These included vegetation changes, soil quality and compaction, and local hydrology. Furthermore, a comprehensive qualitative technique – Landscape Function Analysis (LFA, Tongway & Hindley 2004) – was also incorporated. LFA is designed to determine rangeland health through a wide range of rapidly assessed variables. It provides three indices which together provide a measure of landscape function at each site. These three indices are; Soil Stability, Nutrient Cycling and Infiltration. Furthermore, by determining if such a technique was accurate and viable in this area, it could provide a useful tool through which the Okhombe monitoring group may be able to demonstrate the delivery of specific ecosystem services. Such capability is an important component of a true Payment for ecosystem services (PES) programme (Wunder 2005). Furthermore, if the OMG (Okhombe monitoring group) were easily and accurately able to monitor key factors encompassing soil function, hydrology and vegetation dynamics, they would better understand the impacts that their restoration efforts are having, and tailor their techniques accordingly. These variables were compared not only on open rangeland and adjacent exclosed, restored plots, but also on a relatively pristine plot within Royal Natal National Park, in order to establish baseline values on land of good condition. By comprehensively assessing these variables, and also examining LFA, this study aims to not only integrate the monitoring of ecosystem processes, but to also investigate a new tool which may assist with the ground breaking work being carried out by the OMG.

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1.2. Integration of general ecological theory and site specific variability

General ecological theory helps to explain the various mechanisms and complex interactions taking place within ecosystems. The application of such theories at a given restoration site is crucial to understand the processes at play, and to plan restoration actions accordingly. For example, Grimes competitive, stress-tolerant, ruderal (CSR) approach suggests that three archetypal plant strategies have evolved to cope with two major limiting factors – stress and disturbance (Grime 1977). Although the intensity of, and the response to these factors varies along a gradient, for simplicity's sake only the extremes are described. For environmental conditions where both stress and disturbance is low, a competitive strategy where plants invest most of their energy into overcoming their neighbours is best suited. Under levels of low stress and high disturbance, a short lived, ruderal strategy where plants can rapidly reach maturity and propagate between disturbance events is most advantageous. And finally, under high stress and low disturbance, stress tolerant life strategies would be most successful, as plant densities seldom reach levels where competition effects become manifested (Grime 1977). Such theoretical frameworks can assist in the selection of species for restoration based on traits which are better adapted to local conditions (Grime 1977; Whisenant 1999; Chaplin 1993). Overarching theory also highlights physiological and environmental constraints which may be important when considering a particular restoration action. In the prior example, the theory emphasises the inevitable trade-offs associated with different life history strategies – which in turn translate to direct tradeoffs when choosing restoration measures. A local application of Grimes theory in selecting species for restoration could be used to support the planting of *Paspalum notatum* swathes in Okhombe. This exotic grass species may be ideally suited to stabilising easily eroded soils, as they are graze-tolerant and anchor soils by forming a dense stoloniferous mat. However, such a decision would have to be carefully weighed against the negatives of utilising a non-sterile exotic grass, such as the exclusion of native species.

Despite the importance of embedding site specific planning within broad ecological theories, such an approach is often inadequate to provide the context dependent information required for on the ground management decisions and choices. In such situations, it is the exceptions, rather than the rules that pose the greatest challenge. This is of particular importance in the field of restoration, and if such site specific issues are not taken into account, restoration efforts are often unsuccessful (Wassenaar et al. 2007).

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Some of the most important site specific considerations are the different processes which occur between vegetation and the underlying soils. These plant-soil interactions are important to consider when planning restoration, as poor soil conditions may severely constrain plant performance and growth (Pywell et al. 2003), so much so, that restoration attempts may well fail altogether if such limitations are ignored. Plant composition also influences almost all aspects of soil structure and function, and plants are often used as a tool through which to ameliorate poor soil conditions and facilitate revegetation (Eviner & Chapin 2001). However, this bi-directional feedback can prove to be a curse or cure, either changing soil conditions to allow native vegetation to return, or reinforcing conditions which hamper recolonisation. For example, alien invasives may dramatically alter soil properties to suit only themselves (Ehrenfeld et al. 2005; Vinton & Goergen 2006; Levine et al. 2006).

1.3. Passive (spontaneous) vs. Active (technical or assisted) restoration

The community based restoration practices conducted by the Okhombe Monitoring Group are a form of active or assisted restoration, where direct actions are implemented to facilitate the recovery of vegetation. This is opposed to passive restoration, where the area is simply left to regenerate unaided. Active and passive restoration are not incompatible however, and both approaches should be viewed along a spectrum between high intensity (and typically also high cost) technical intervention or a completely "hands off", passive approach (Prach & Hobbs 2008). However, spontaneous succession may interfere with technical intervention to varying degrees, and vice versa. In a recent review, Prach & Hobbs (2008) suggested that finding the most balanced approach for a given restoration project depends on three key aspects. The goal of the restoration project, the stress/productivity levels of the site, and the proximity of the site to propagule sources.

Sites which experience moderate stress and productivity are likely to possess the greatest diversity (Grime 1977) and therefore have a greater species pool upon which to draw from when re-colonising degraded lands, thus facilitating a passive approach. Conversely, technical assistance may be required for sites on either end of the stress/productivity gradient, as they are more prone to the dominance of a few species (Lonsdale 1999; Huston 2004). Areas which are extremely stressed or of very low productivity will likely require

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considerable technical intervention to reach a stage where autogenic recovery is possible (Whisenant 1999).

Another key driver of spontaneous succession proposed by Prach and Hobbs (2008) is propagule pressure and proximity. Work done in a variety of ecosystems (including grasslands) by Rehounkuva & Prach (2007) found that at distances greater than 100 m from a propagule source most species were unable to recolonise a disturbed site. Evidence suggests that in the sourveld (a South African grassland region of typically lower palatability, where the study site is located), dispersal distances are even smaller. *Themeda triandra*, one of the most valuable palatable grasses in the region, was found to disperse less than 2 m from parent plants, with extremely poor seedling success (Everson et al. 1985). This highlights the importance of the surrounding vegetation when considering spontaneous restoration. In the situation at Okhombe and many communal rangelands in South Africa, large contiguous areas have been subjected to intensive grazing for long periods of time. Consequently, there are few nearby propagule sources which can recolonise an area, should it be excluded from grazing pressure. This lack of propagule pressure, coupled with the low germination success suggests that technical intervention may be necessary in order to achieve restoration goals in this area.

While technical intervention is often necessary for the restoration of severely degraded lands, passive restoration, or spontaneous succession does possess a number of benefits. Spontaneous succession is usually rapid in highly productive areas (Walker & del Moral 2003), meeting the most common goal of restoration, which is rapid revegetation to secure soil loss or enhance aesthetics. Passive restoration is typically far less costly than intensive or active restoration. This is an advantage as financing for restoration projects is often insufficient. Apart from simply being more cost effective, spontaneous revegetation allows for the colonisation of well adapted local species (Kovar 2000), often resulting in higher natural value, and providing better refugia for wildlife (Hodacova & Pach 2003). However, this process may take considerably longer to reach a target state, especially if the site is large and propagule pressure is weak. Furthermore, if the site is surrounded by ruderal or alien species it is likely to be colonised and dominated by such species. Technical approaches such as physical clearing or engineered soil stabilisation are often able to more rapidly revegetate an area under adverse environmental stresses or low productivity, and also allow greater control over the initial distributions, density, or timing of vegetative growth on the sites.

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While spontaneous succession does offer several benefits over technical intervention, there are situations where it is unlikely to be adequate as a restoration tool. Particularly in the degraded sites around Okhombe, low productivity, lack of propagule sources and poor establishment success of indigenous grasses suggest that some form of intervention is needed in order to facilitate restoration

1.4. Grazing management and restoration

Overgrazing is one of the chief issues facing degraded rangelands across the world, particularly in developing countries (Holocheck et al. 2006). This poor management action may lead to reduced species richness, removal of vegetation cover, replacement of palatable with unpalatable species, exposure of soil and severely increased erosion rates (Thornes 2007). However, for rangelands with a past evolutionary history of large herbivore pressure, removing livestock entirely may result in disrupting natural processes and mechanisms and ultimately an overall loss of biodiversity. In such areas, the structure, function and species composition have all evolved alongside grazing (Papanastasis & Peter 1998; Perevolotsky & Seligman 1998). However, it is important to bear in mind the levels of intensity of historical grazing pressures. For example, the Moist Highland Sourveld which dominates the northern Drakensberg, is nutrient poor due to heavily leached soils, and likely supported only low herbivore densities in the past. Thus, appropriate grazing management should include an adjusted stocking rate which is suited to the capacity of the restored land. In addition the correct livestock species should be stocked, and a grazing system developed which is well suited to ecosystem functioning (Papanastasis 2009). Grazing has a wide array of impacts, affecting both structure and function of rangeland systems. These impacts can be positive or negative, depending on how the grazing is managed. Grazing management as a restoration tool is, however, unable to overcome situations where abiotic conditions have deteriorated significantly. Instead, it should be seen as a potential tool for manipulating biotic conditions, for example reducing cover of certain species, preventing woody encroachment or encouraging growth through the removal or moribund matter (Whisenant 1999, 2002; Hobbs & Harris 2001; King & Hobbs 2006; Papanastasis 2009).

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Both spatial and temporal management of grazing provide avenues in which to selectively control biotic factors on rangelands (Papanastasis et al. 2008). In the nutrient poor Moist Highland Sourveld, this aspect may be of particular importance, as it is unlikely that a confined area would be able to sustain grazing pressure for any considerable length of time. However, even before such considerations are made, vegetation needs to return to a suitable late succession stage, with a diverse representation of climax species. In Okhombe, the vegetation in the restored areas is still far from reaching a climax stage, and given the degree of degradation, may never do so. For this reason, the Okhombe Monitoring Group have not employed grazing as a management option. To better manage communal rangelands in the area, a grazing plan for the communal rangelands in Okhombe was implemented in 2004/2005 (Tau 2005), although it has since fallen out of strict use. A formal grazing management plan could prove a vital step to better managing veld condition through manipulating grazing pressure, and the resurrection of such a system would likely yield promising results.

1.5. Lessons learned from international restoration projects

In France, a long history of extensive engineering and agricultural projects has altered large tracts of once species rich grasslands. This degradation has since been tackled through various restoration approaches (Muller et al. 1998). A review of some of these past restoration programmes allows for the chance to gain a better understanding of how common mechanisms and processes respond to certain restoration actions. This in turn allows for better planning of future restoration efforts, by identifying effective techniques and methodologies.

A state-transition model provides an effective means of illustrating some of the key dynamics in degraded ecosystems (Aronson et al. 1993; Westoby 1989). In such a model, various abiotic and biotic thresholds can be seen as barriers requiring differing degrees of technical intervention or recovery time in order to reach restoration goals. If degradation is slight, passive restoration may suffice for a return to near-natural conditions, whereas more intense degradation will likely require some degree of active intervention to overcome the threshold. If successful, this intervention may lead to a new transitional ecosystem. Over time, this transitional system should reach a steady state, and ultimately grow to closely resemble pre-

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anthropogenically impacted conditions. Lastly, even if a degradation threshold is crossed, it may be possible for that ecosystem to persist and remain stable, albeit within a new “steady state” (Aronson et al. 1993). In light of Grimes Competitive, Stress-tolerant, Ruderal (CSR) approach (Grime 1977) discussed earlier, it is important to note that the two theories are not incompatible compatible, but rather should be viewed as one nested within the other. For example, competition, stress tolerance and ruderal strategies may be adopted by plants which exist within a greater environment controlled or bounded by various thresholds, which, if overcome, will shift the advantages or disadvantages of differing life history traits.

Once a field has been abandoned following grazing, it is often quickly overtaken by robust, fast growing species (first herbaceous, but later woody) (Prach 1993). The removal of disturbance disadvantages poor competitors and ultimately results in their loss due to competitive dominance (Alard et al. 1994). In such situations, the reintroduction of a controlled stress which mimics more natural conditions may allow for the weaker competitors to return. Such interventions may employ domestic herbivores (Gordon et al. 1990), and other forms of stress such as mowing, cutting and burning.

However, if an agricultural or engineering practice has drastically transformed a landscape, for example levelling of slopes, or the removal of topsoil, often technical interventions are required. It may be necessary to provide a foothold for desirable species to establish, and this may require removal of vegetation, alteration of topsoil, providing a source of propagule pressure or other interventions (Berendse et al, 1992). Often after particularly large scale or intensive degradation, for example post mine closure or after large engineering projects, the most crucial function to establish is stabilisation of the soils. In such cases the creation of an intermediate steady state, such as an artificial meadow sown with suitably adapted species may provide a solution. These “ecosystems of substitution” may then be progressively replaced with indigenous plant communities. In this manner, alternate steady states can provide a “stepping stone” on the road to restoration – by preventing soil loss and associated long term impacts. (Bédécarrats 1991, Muller et al 1998).

In Okhombe, an alternative steady state model may provide a sound conceptual mechanism for the domination of heavily grazed areas by tough, stoliniferous lawn-forming grass species, particularly *P. notatum*. Observational evidence suggests that these lawns are characteristic of degraded areas, as they are largely absent in pristine areas with low

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herbivore pressure. In degraded areas in Okhombe, such a transitional system is potentially desirable however, as the dense stoliniferous lawns may help to bind soils and arrest erosion. Whether these stoliniferous lawns can act as effective “substitute ecosystems”, and are capable of ultimately returning to some semblance of pre-disturbance ecosystem structure remains to be seen, and long periods of rest or active intervention may be necessary.

1.6. Livestock and the impacts of trampling on soil hydrological characteristics

Although many of the world’s rangelands evolved under the pressure of grazing ungulates, the rearing and maintenance of domestic livestock may have negative impacts on rangelands, as a result of artificially increased herbivore density (Warren et al, 1986).

Studies have long shown that maintenance of domestic livestock impacts both biological factors such as composition & cover (Ellison 1960) and soil physical properties (Reed & Peterson 1961). Generally, infiltration tends to decrease under increased stocking pressures, whereas run-off and sediment loss are exacerbated (Rauzi & Hanson 1966, Rhoades et al. 1964). In a study by Warren et al (1986), it was shown that trampling in itself (i.e. on bare soil, uncomplicated by other factors such as vegetation removal) resulted in reduced infiltration and increased soil loss with increasing stocking rates. Particularly under heavy grazing pressure, soil and hydrological characteristics tend to be negatively impacted, while moderate to light grazing intensities are substantially less harmful and are often not significantly different from one another (Gifford and Hawkins 1978; Van Poollen and Lacey 1979).

These effects tend to be even more pronounced on wet soils than on dry soils. Infiltration rate decreased and sediment production increased significantly when plots were trampled when moist, as opposed to those which were trampled when dry (Warren et al 1986, Edmond 1962, Lull 1959). Soil bulk density was found to be a good predictor of infiltration rate and sediment production (Warren et al 1986). This relationship was noted in other studies, where increased stocking rates led to greater soil bulk density, lower infiltration rates and greater sediment production (Knoll & Hopkins 1959, Rauzi & Hanson 1966). In Okhombe, sediments are washed down slope into local streams, resulting in the removal of nutrients and increased sedimentation of dams.

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1.7. *Biological soil crusts, and their role in ecosystem function and restoration*

Biological soil crusts (hereafter biocrusts) occur as a thin horizontal layer on the soil surface composed of lichen-bryophyte and microbial communities. Biological soil crusts differ fundamentally from physical crusts in terms of the processes they facilitate. While both engender a degree of stability to the soil surface (Uchida et al. 2000), biological soil crusts may provide further positive benefits which are often absent with physical soil crusts. Such benefits include enhanced aggregation of soil particles (Mazor et al. 1996), increased infiltration (Brotherson & Rushforth, 1994), and increased soil fertility through carbon and nitrogen fixation (Evans & Ehleringer 1993; Lange et al. 1994).

Biocrusts play a wide and varied role in ecosystem function (Bowker 2007). Particularly, they are recognised as an important early sere in primary succession of terrestrial plant communities, (Rayburn et al. 1982; Kurina & Vitousek 1999). Over time, biocrust cover may recede as other vegetation establishes, but they are seldom completely removed. Rather, they persist in lower densities, and exploit opportunities in secondary successions (Bowker 2007). Although they are most common in areas where a high proportion of light reaches the soil surface, they are found in almost every ecosystem in at least some form or successional stage (Mando et al. 1994).

In high stress environments, biocrusts are often a more permanent and important feature. If removed, the consequences may be widespread and interlinked – sometimes even acting as a trigger between alternative steady states (Miller 2005; Yates & Hobbs 1997). Mechanisms by which such a shift may occur include the loss of erosion resistance, microbial activity and nutrient fixation (Maestre et al. 2005; Belnap 1995), and changes in water distribution (Eldridge et al. 2002). Although biocrusts in high stress environments are often quite vulnerable to disturbance, passive restoration efforts such as exclusion of any disturbances to the soil crust may enable recolonisation and recovery (Anderson et al. 1982). However, passive recovery does not always occur (Belnap & Warren 1998), and considerable intervention may be required if a state transition has occurred with the loss of the biocrusts.

Although widely recognised as important ecosystem features, and receiving considerable

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academic attention (Belnap & Lange 2003), biocrusts are under-represented in the restoration literature. A meta-analysis of every article published in the journal *Restoration Ecology* from 1996 to 2006 found that less than 2% of the papers presented some form of data on biological soil crusts (Bowker 2007). This may be due in part to often very slow recovery times following disturbance, making it difficult to incorporate biocrusts in restoration actions. Observation and estimates have suggested that recovery times may range from six years (Belnap & Eldridge 2003) to as long as centuries or even millennia (Belnap & Warren 1998). Despite this, there is some evidence that assisted recovery can result in greatly reduced return periods (Grettarsdottir et al. 2004; Li et al. 2004), and such actions may prove a useful addition to restoration practices.

One such assisted recovery procedure is the stabilisation of soils. Artificial soil stabilisation through application of course litter or the establishment of stabilising vascular plants has yielded considerable successes in the restoration of biocrusts (Fearnehough et al. 1998; Danin et al. 1998). Although the recovery of biocrusts was not a direct goal in the restoration work at Okhombe, soil stabilisation activities were carried out at all sites, and should contribute to more rapid recolonisation of biocrusts in these areas.

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1.8. Soil quality, soil organic matter, and their impact on ecosystem function

Soil quality is widely recognised as one of the most important aspects to be addressed in restoration interventions, as it impacts on so many other factors (Jordan et al. 1987; Heneghan et al. 2008). Although soil quality can vary in many different ways, and be influenced through any number of land use practices it is often only expressed in terms of soil loss through erosion. While erosion is undoubtedly one of the most serious aspects of declining soil quality, it is important to also consider other ways in which soils can be negatively affected (Mills & Fey 2003). For example, in South Africa, widespread degradation of soils was first addressed with the formation of the Soil Erosion Advisory Council in 1930, and later the Soil Conservation Act in 1946. While these actions aided in reducing soil loss in many areas across the country, they failed to address other aspects of declining soil quality – such as changes in the physical and chemical properties of soils due to land use practices (Mills & Fey 2003).

One of the most useful indicators of soil health using chemical and physical properties is soil organic matter (SOM). Measured primarily by organic carbon content, and usually paired with nitrogen content, SOM provides a robust and meaningful indicator of soil health, particularly due to the role of SOM in nutrient cycling (Mills & Fey 2003). For example a general trend of decreasing SOM with increasing cultivation duration and intensity has been reported in several South African studies (DuToit et al. 1994; Nel et al. 1996.). A strong, linear relationship has been demonstrated between veld condition, basal cover and SOM content by Du Preez & Snyman (1993). They found that over three sites, SOM decreased with lower basal cover and veld condition. This was attributed to significantly lower biomass production and greater soil temperature on the poorer quality sites. Lower biomass production reduces the amount of organic matter being returned to the soil, while greater soil temperatures may increase microbial activity and decomposition.

Soil erosion does play a critical role in the loss of SOM, as SOM decreases exponentially with depth in the soil profile (Woods 1989). Thus, losing only the first few centimetres of soil can have a disproportionate impact on SOM. Vegetation loss also typically reduces SOM, although the process is somewhat more complex. Not only are the direct inputs of litter and root biomass lost with devegetation (Allsopp 1999), but other indirect effects may manifest. These include both positive and negative changes to; microclimate temperature and moisture (Du Preez & Snyman 1993; Savage & Vermeulen 1983), plant root influence on microbial activity and nutrient cycling (Theron 1951). The absence of vegetation also favours the formation of physical soil crusts, as raindrops possess greater momentum and impact (Allsopp 1999). This may act synergistically with reduced SOM to increase crusting, as reduced SOM can reduce aggregate stability and increase clay dispersion (Cook & Dalal 1992; Tisdall & Oades 1982, Blair & Crocker 2000). However, as discussed earlier, biocrusts may colonise these physical soil crusts, which may ameliorate low infiltration rates (Brotherson & Rushforth 1994). It is important to note that physical soil crusts are not exclusively disadvantageous. Sometimes they can provide significant benefits to ecosystem function, such as protection from both water and wind erosion (Sombroek 1986). However, they may also impede vegetative growth by reducing water infiltration, lowering oxygen levels in the pedoderm, restricting root growth, and decreasing seedling germination and growth (Shainberg & Levy 1994; Bristow 1988).

Soil organic matter tends towards an equilibrium state over time, acting as a finite sink rather

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than a unidirectional process (Jenkenson & Rayner 1977 ; Paul et al. 1997a). Although the influence of different land use practices on total SOM is often varied and unpredictable, some are clearly associated with a certain outcome. For example, the cultivation of fields (unless extensively fertilised with manure) almost always reduces the amount of soil carbon (Francis et al 2001; Edwards et al 1992). Rangelands are also easily prone to declining soil quality. High densities of livestock remove vegetation through grazing, reducing SOM and other nutrients (Allsopp 1999). Given the extent of such land uses, it is clear that declining soil quality is an important issue even without considering the effects of soil erosion.

However, it is not only anthropogenic factors that influence soil quality, climate is also an important influence on SOM. Previous work in the South African Highveld found that SOM was lost more rapidly in warm, dry areas than cool, wet areas (Du Toit & Du Preez 1995). Fires too, play a critical role in the amount of SOM in soils. Generally, fires tend to reduce SOM primarily through changes in soil chemistry, as well as through direct losses such as ash and reduced vegetation cover (Mills & Fey 2003; Seastedt 1994).

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But how does one address the issue of poor SOM content? Once soil quality is reduced to the point that plant growth and germination is severely hampered, a shift in "ecosystem steady state" may occur, and returning degraded lands to a more functional state may be very difficult to achieve (Du Preez & Snyman 1993). Increasing productivity and preventing removal of vegetation within a given area returns greater quantities of carbon to the soil. This may allow for soils to be used as a carbon sink which can be utilised by enhancing the natural processes of vegetation productivity and litter decomposition. One of the most prominent points that emerge from the literature is the critical role that vegetation cover plays, and its preservation or restoration should be a primary mandate when managing soil quality (Mills & Fey 2003). This is a common goal of many restoration programmes, and the actions taken within Okhombe to ensure that vegetation is not grazed on restoration sites will also aid the recovery of SOM and overall soil quality.

1.9. Carbon sequestration and storage in grasslands

The relatively recent realisation of the extent to which anthropogenic impacts are altering the global carbon cycle has generated newfound interest in understanding how these cycles operate. This direction gained even further attention since the recognition of carbon sequestration by soils as a means by which countries may reduce their total carbon budgets (IPCC 2001). Particularly, research has been driven towards identifying important carbon pools and how they can be influenced so as to sequester greater amounts of atmospheric carbon, in order to ameliorate that released by anthropogenic activities (Prentice et al., 2001; IPCC, 2001).

In terrestrial carbon pools, the vast majority of carbon is stored below ground. This is particularly true for grassland ecosystems, where as much as 98% of total carbon exists in subterranean pools (Hungate et al. 1997). Although these subterranean pools are generally subject to much slower turnover rates than above ground stores (Jenkinson, 1990; Paul et al. 1997b; Post & Kwon, 2000), they are still prone to dramatic losses when processes such as increased soil erosion and land degradation occur (Kalbitz et al. 2000). As mentioned previously, this is in part due to the fact that the bulk of soil carbon is stored in the top layers of the soil profile (excluding ancient reservoirs which have been buried and transformed for vast periods - such as oilfields or coal deposits). For example, in a study looking at temperate grasslands, 75-80% of root biomass (an important contributor to soil stored organic carbon) was found within the first 30 cm of the soil profile (Reeder et al. 2004).

Four main factors determine the amount of carbon sequestered by grasslands; the rate of input of organic matter, how easily decomposed the organic matter inputs are, the depth at which the organic carbon is stored, and how well protected stored organic carbon is from breakdown (Jones & Donnelly 2004). Soil organic carbon is protected from microbial breakdown when its fractions are coated with clay and other mineral particles. These coated particles are then further sealed within soil micro and macro-aggregates, through a chemical binding process (Post & Kwon 2000). These processes are typically interrupted when grasslands are converted to agriculture, and comparison between grasslands and croplands have found significantly reduced carbon sequestration in the latter (Conant et al. 2001). This reduced carbon capture is due to a range of factors. Removing vegetation results in lower carbon inputs from standing vegetation (Jackson et al, 1996) as do the associated secondary effects discussed earlier (Mills & Fey 2003). Disturbance and exposure of the soils through tilling (Jones & Donnelly 2004) and particularly decreased protection of soil organic carbon from breakdown, also reduces soil-stored carbon (Gregorich et al. 2001).

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Research comparing soil texture and carbon storage suggests that the capacity of a soil to hold carbon is directly related to the proportion of silt and clay found in the soil (Hassink, 1997). Soils in Okhombe are high in clay content, suggesting that they may have a high carbon storage capacity. However, inherent soil properties of a given site cannot be easily influenced through restoration action, and while such information may help to pre-select areas that are best suited for enhanced carbon sequestration, recommended management actions must provide on-the-ground guidelines for restoration. In broad terms, management which focuses on reducing disturbance to the soil profile (such as no-till agriculture or removing trampling livestock) and increased standing biomass (such as resting heavily utilised land, or introducing year-round crops) are possibly the most effective means through which to enhance carbon sequestration (Guo & Gifford 2002). The difference in carbon storage between permanent grasslands and utilised land illustrates the capacity by which carbon sequestration can be enhanced. Conversion from grassland to arable croplands was found to reduce soil carbon by almost 60% (Guo & Gifford 2002). Jones & Donelly (2004) suggest that grasslands which have been poorly managed present the greatest opportunity for soil carbon storage, as stores are effectively "empty" and available to be filled. In this sense, degraded soils can be seen as an asset for carbon storage, since utilisation as a carbon store will have multiple positive impacts. Not only as a source of carbon sequestration, but also as an opportunity for restoration, with the associated benefits of enhanced ecosystem services and function.

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1.10. Determination of hydraulic properties from tension and double-ring infiltrometers.

Determining soil hydraulic properties allows for an understanding of the different hydrological functions and processes which occur below the soil surface. This in turn can provide insight into the impacts of both natural and anthropogenic influences on water flows and associated ecosystem services. It may also allow for a broader understanding of landscape scale hydrology, from available water flux for vegetation to the infiltration of point source pollution into ground water (Angula-Jaramillo et al. 2000).

Certain hydraulic properties lend themselves as indicators of greater, landscape level processes and functions. One of the most oft-measured and important hydraulic properties is infiltration, or hydraulic conductivity of soils (Angula-Jaramillo et al. 2000). Infiltration is a measure of how well water is able to permeate into the soil, and thus provides an idea of how much water is available to vegetation following precipitation. Importantly, reduced infiltration results in greater surface runoff, transporting the water away from the soils where it falls. This does result in increased overall water yields, as less water is transpired through vegetation (Hibbert 1983), however base flows are often severely reduced in the dry season, as soils are no longer able to capture and store local precipitation (Allan 2004). This is relevant not only to the persistence of vegetation, but also to local communities who are dependent on such local flows during the dry season (Everson et al. 2007).

1.11. Applying ecological theory and a multi-variable approach to the Okhombe project

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By taking into consideration multiple processes which influence rangeland health, it should be possible to better understand how a given restoration action will impact local site conditions and vice versa. It is important not only to reconcile the many different factors and processes at play which impact vegetation dynamics, but also to integrate various ecological theories which govern our understanding of how these processes operate and interact (Briske et al. 2005). Such knowledge is critical to restoration and any form of rangeland management (Dyksterhuis 1949). Historically, rangeland management was dominated by approaches which are based upon a single predominant theory – such as succession (Clements 1916). This leads to inevitable confusion when processes are encountered which do not fit the chosen model (Briske et al. 2005). For example, non-linear processes such as episodic weather events, changes in fire regimes or severe erosion are able to alter vegetation dynamics in a manner that contravenes traditional succession theory (Westoby et al 1989). By integrating multiple ecological theories, for example accepting that traditional succession can occur within a framework of multiple stable states, it is possible to explain and understand seemingly conflicting rangeland dynamics (Briske et al. 2005). Furthermore, such a framework allows for the integration of multiple factors such as soil quality, infiltration or vegetation dynamics and the understanding of their synergistic outcome (Westoby et al. 1989; Briske et al. 2003). In this study, the evaluation of key indicators of rangeland health such as species composition, vegetation cover, infiltration and soil quality is couched within a framework of integrated ecological theory. This should allow for a comprehensive understanding of how restoration actions are influencing the recovery of degraded areas in Okhombe. In areas where degraded areas have long surpassed several thresholds after extensive utilisation, it is important to focus on the restoration of ecosystem functions, rather than a succession-stage semblance of species composition (Stringham et al. 2003). In this regard too, the choice of assessing a wide range of process orientated variables is well suited.

1.12. Measuring restoration success

The Society of Ecological Restoration International recommends nine ecosystem attributes as indicative of restoration success (SER 2004). They suggest that a successfully restored ecosystem; (1) has similar diversity and community structure to a comparative reference site, (2) includes the presence of indigenous species, (3) possesses all functional groups necessary for long term stability, (4) is able to sustain reproductive populations, (5) has normal functioning of ecological processes, (6) is properly integrated with the surrounding landscape, (7) has removed potential threats to ecosystem health, (8) is resilient to natural disturbance, and (9) is self sustainable (SER 2004). A comprehensive literature review has shown that measuring all of these variables accurately is far beyond the scope, time-line and funding of all but the most comprehensive restoration assessments (Ruiz-Jaen & Aide 2005). Instead, it was recognised that a more realistic approach was for variables to be chosen to reflect some measure of three key ecosystem attributes, namely diversity, vegetation structure and ecological processes (Ruiz-Jaen & Aide 2005). Furthermore, the importance of including reference sites through which to determine restoration targets, was highlighted (Ruiz-Jaen & Aide 2005). In order to follow this approach, it was highlighted that this study must measure several indicators of ecosystem processes across these three categories, and a benchmark site in a conserved area (in this case, Royal Natal National Park) should be assessed to provide a meaningful comparison.

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2. Chapter Two.

The ecological Impacts of community based restoration on communal grasslands in the Drakensberg foothills

2.1. Abstract.

A community based restoration programme has been in place for over a decade in the ward of Okhombe in the Northern Drakensberg. This area is an important water supply region for South Africa, and has been identified as a potential site for the implementation of a Payment for Ecosystem Services market. This study aims to quantify some of the ecological impacts of community based restorations, and to investigate the use of indicators to measure the delivery of ecosystem services. Several indicators, both biotic and abiotic were used to examine multiple processes and factors which contribute to rangeland health, including Landscape Function Analysis. Overall, both degraded and restored plots within Okhombe were found to be well below the level of ecosystem health and function of a reference site in a nearby conservation area. Average infiltration rates were consistently higher on restored plots, indicating that restoration action has enhanced this important ecosystem process. Basal cover revealed the dominance of large swards of the exotic grass *P. notatum* on many sites, which was more prevalent on the open, grazed rangeland than on restored sites, and absent from the reference site at Royal Natal National Park. However abiotic indicators did not show any positive effects associated with the increased basal cover, highlighting the importance of using both biotic and abiotic indicators. When compared to the reference site in Royal Natal National Park, all sites (both restored and open rangeland) within the communal rangelands showed poor veld condition and grazing capacities. Previous literature has suggested that active management of grazing pressures and reintroducing a biennial spring burn regimes coupled with appropriate rest, is needed to promote palatable species. Landscape Function Analysis was shown to be unsuitable for the evaluation of restoration success in this study, with no significant correlation between its qualitative indices and traditional quantitative data. This study was able to provide important insight into the use of several different indicators of ecosystem function, and their application to a payment for ecosystem services programme, and to furthermore establish a baseline for future research.

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2.2. Introduction

The Maloti-Drakensberg is the largest mountain range in South Africa, reaching heights of almost 3,500 m above sea level in places. Also known as "uKhahlamba", or the "Barrier of Spears" it creates a formidable barrier between South Africa and the Kingdom of Lesotho (Sycholt 2003). Its core region (the Drakensberg Alpine Centre) contains one of the few World Heritage sites established for both cultural and biodiversity value. It owes this status to the presence of more than 40,000 Khoisan rock art paintings, and 2,520 species of higher plants, of which 13% are endemic (Carbutt & Edwards 2006). Despite occupying only 5% of the national land surface, the Drakensberg is arguably South Africa's most strategic fresh water supply, providing a quarter of the national demand by means of extensive river and inter-basin transfer networks (Diederichs & Mander 2004).

Although 24,200 km^2 of the mountain range and its surrounds falls under official protection as a World Heritage site, it is bounded by a roughly equal area which receives no formal protection. This surrounding unprotected area is subject to a range of impacts from rural and commercial agricultural practices and development (Cowling et al. 1994). Furthermore, these unprotected areas lack any sort of formal management plan, and inappropriate land use practices such as overgrazing and incorrect burn regimes may lead to vegetation loss, increased runoff and soil erosion (Everson et al. 2007). In steeper areas and foothills increased erosion manifests as large "dongas" – distinct gullies carved out of the slope by surface runoff and contraction and expansion cycles caused by frost. These dongas result in the loss of productive land for local communities, both for grazing of livestock and growing crops, and at times even threatening to undercut homesteads. Altered quality and quantity of runoff from these areas may lead to reduced base flow and water availability in the dry season, and greatly increase sediment loads.

Okhombe is a rural community on the outskirts of the protected area, home to approximately 4000 inhabitants with roughly 4000 cattle and 2000 small stock, mostly goats (Everson et al. 2007). The community relies heavily on nearby natural resources for daily life. For example, cattle graze the hill slopes in summer, and people utilize water from local catchments, and grow crops in the valley floor (Everson et al. 2007). However, as a result of degradation and loss of vegetation, local streams wash silt down from the Upper Thukela catchment into the

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dams that form the important Tugela-Vaal water transfer scheme (Everson et al. 2007). This can necessitate costly action including the physical removal of silt through dredging, or over-engineering of dams. The over-utilisation of these communal rangelands can also result in the loss of soil carbon storage and reduced ability of soils to sequester carbon (Knowles et al, 2008). Furthermore, reduced base flow means that less water is available for direct abstraction to local communities in the dry months. This is particularly problematic as no piped water utilities have as yet been supplied by government to the community. Abstraction from local water sources and several well points scattered throughout Okhombe provide the only available water to the community.

The Drakensberg Mountain Range has been identified as a priority area for the development of new markets based on ecosystem services and natural capital (Blignaut et al. 2008). This is due to a combination of both opportunity for development, and potential loss of valuable ecosystem services. This includes high levels of ecosystem productivity, value as a water source, extreme levels of poverty, poor infrastructure, and importance to biodiversity. Payment for ecosystem services (PES) is one such alternative market system, wherein land owners and users are remunerated for changing land use practices such that valuable natural processes and goods are protected. In this way PES provides a novel approach to conservation, by assigning a value to the continued health of a given environment, and attaching direct monetary value to the service it provides (Wunder 2005; Daily 1999). South Africa has a well-documented and successful past record with PES-based programmes (Blignaut et al. 2008; Marais & Wannenburgh 2008; Turpie et al. 2008). One of the most notable thereof is the Working for Water initiative, which has served as a model for a number of state funded PES programmes (Turpie et al. 2008). The implementation of a PES programme aimed at rehabilitating degraded catchments has the potential to provide greater economic returns than conventional (engineered) water development programmes, and furthermore safeguards against environmental degradation and loss of ecosystem services (Blignaut et al. 2008). Such a project would generate incentives for land users to change land use practices in a manner that would enhance the output of ecosystem services. These changes would include measures such as restoration of degraded catchments, maintaining livestock at lower densities, practicing rotational grazing, changing current annual winter burning practices to biennial spring burns and avoiding summer burns (Blignaut et al. 2008; Uys et al. 2004,). If such a project was fully realized in the area, it could provide substantial employment opportunities for local communities, along with skills transfer and training and

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provide a means of escape from poverty.

The groundwork for such a programme is already in place within Okhombe, and some of the benefits are already being seen. A governmental LandCare project was launched in Okhombe in 1999 with a focus on job creation through the restoration of degraded areas in the Drakensberg catchment area. The LandCare programme encouraged land users to take on the responsibility for local issues and tackle these problems themselves. Through the LandCare project, capacity building and training oversaw the instruction of community members in a number of different erosion control techniques. Both physical measures (such as the packing of stones, mattressing, digging of swales and creation of cattle steps), and vegetative restoration techniques (planting of Vetiver grass, trees in micro-catchments and indigenous and sterile exotic grasses on degraded slopes) were used at various sites with reported success (Everson et al. 2007, Sistika 2004 in Everson et al. 2007). Swales are fairly deep ditches (approx 60cm) dug across an area, parallel to contour lines, such that any surface runoff is arrested. Cattle steps constructed along steeper access routes, where logs are places perpendicular to the track in order to prevent erosion and arrest soil lost. The sterile Vetifer exotic grass used in Okhombe is an exotic grass species with exceptionally deep root structures capable of binding the soil below it, typically planted parallel to contour lines in order to arrest sediments. However, there was a lack of concrete evidence to quantify the degree of success achieved by these restoration activities (Everson et al. 2007). To address this issue a progressive community based monitoring system, funded by the Water Research Commission, was implemented at Okhombe where participating members of the local rural communities were trained to perform scientific measurements and analytical techniques (Everson et al. 2007). The acquisition of technical, scientifically based monitoring techniques by a small, rural community represented a considerable challenge, considering that many of the participants lacked formal education. Despite this, the project was successfully implemented through a community based and led organisation - the Okhombe Monitoring Group (OMG). The OMG was formed by representatives from each of the community's six sub-wards, and included a number of volunteers who would assist with monitoring and restoration. Notably, both groups agreed to work on a voluntary basis with no pay, and only recently have funds been allocated to remunerate them for years of effort and restoration services (T. Everson *pers comm*). Key to the success of this project was the development of simple, informative monitoring techniques. Measurements taken included rain splash height, up/down slope erosion, donga profiling, sediment deposition, basal cover, precipitation, and

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water quality and quantity (Everson et al. 2007). Through both the Landcare and the OMG monitoring projects, a progressive, integrated approach to community based natural resource management was demonstrated to the local community. The importance of enhancing ecosystem function by restoring soils, vegetation & natural water flows was made evident through their own observations. This information was in turn distributed to local community members through feedback sessions with the OMG (Everson et al. 2007), further strengthening the local initiative with community buy-in.

The implementation of a payment for ecosystem services (PES) strategy in the region can potentially ensure the effective management of natural resources, whilst safeguarding biodiversity and meeting conservation goals. Restoration of natural capital by local community groups has led to the transfer of advanced skills (including the application of restoration techniques, community trust management and computer literacy), and more recently following buy in from the Working for Water program, a form of income for dedicated community members (Everson et al. 2007). While a true PES programme will pay for a direct service, such as reduced soil erosion or increased infiltration and base flow, the current pilot programme in place at Okhombe demonstrates the potential such markets hold. The full implementation of a PES strategy will require extensive monitoring and certification of the delivery of water services. The community monitoring project at Okhombe provides an ideal platform upon which to build further understanding and skills, and records kept by the group may provide essential baseline information on the impacts of restoration on soil erosion and water quality. However, there remains a need to calibrate the data collected by the OMG, and for it to be verified against more conventional scientific measurements (Everson et al. 2007).

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One of the critical facets of a true PES system is that of conditionality – the business principle that payment is only made if the service is actually delivered (Wunder 2005). One of the challenges with such a principle is measuring the delivery of the ecosystem services themselves. Landscape Function Analysis (LFA) is a technique which allows for the rapid assessment of the health of different landscape processes using easily measured field indicators (Tongway 2003). It serves to complement traditional existing procedures such as species composition and abundance, and adds a functional link to vegetation structure and organisation. Ultimately it has been shown to provide an effective tool to assess how restoration changes landscape function with regard to stability, infiltration and nutrient cycling, at both hillslope and small patch scales (Tongway & Hindley 2004) and may provide an effective, easy to use means of gathering critical data in order to ensure that goods and services are being delivered.

For this paper, the term restoration will be used in preference to restoration, and shall encompass the repair of processes, productivity and services to those of pre-existing ecosystems (full, partial or ongoing). Although in the strictest sense, the term "restoration" implicitly suggests a complete re-establishment of prior species composition and community structure (Aronson et al. 1993), its use in the literature is far more general, and has become part of the standard jargon of the field, so much so, that the field itself takes its name from the term (SER 2004).

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2.3. Methods

2.3.1. Problem Statement

Community based restoration and land management in the Maloti-Drakensberg has the potential to provide greater returns on investments than conventional engineered water storage solutions (Blignaut et al. 2010). Furthermore, it can improve veld condition and water security for local communities, whilst conserving natural systems. A payment for ecosystem services market provides the ideal platform through which to make such initiatives sustainable (Blignaut et al. 2008; Turpie et al. 2008). However, before such a programme can be put into action, a thorough investigation and quantification of the actual benefits derived from restoration action must be conducted (Blignaut et al. 2010). Understanding the ecological impacts of restoration forms a critical component of such an analysis. Community based monitoring has been shown to be effective for local community application (Everson et al. 2007). It can reveal broad trends following restoration, and identify best suited techniques for a given scenario, but verification by more conventional scientific measurements is needed (Everson et al. 2007). However, each case must be evaluated on its own merits.

Hypothesis

The key hypothesis which was tested in this study was that community based restoration of natural capital significantly improved degraded rangelands in Okhombe, thereby improving land productivity, soil quality, vegetative cover and composition and increasing the rate of infiltration *(non-statistical).*

2.3.2. Main Objectives

The main objective of the study was to measure the impacts of the Okhombe community based restoration programme. This was achieved by assessing the changes in vegetation cover, species richness and diversity, veld condition and grazing capacity by comparing these variables in restored, relatively undisturbed and degraded grasslands. A second objective was to measure the differences in infiltration rates, compaction and soil nitrogen and carbon content and to compare these quantitative measures with those derived from Landscape

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Function Analysis (LFA). Finally, the potential for using LFA as a monitoring tool for community-based restoration programmes in Northern Drakensberg Highland Grassland vegetation type (Mucina & Rutherford 2006) was also investigated.

2.3.3. Research Design

2.3.3.1. Study site

The ward of Okhombe occupies a horse-shoe shaped valley in the foothills of the Northern Drakensberg Mountains and falls within the Upper Thukela catchment area in the KwaZulu-Natal province of South Africa (28°42’ S; 29°05’E).

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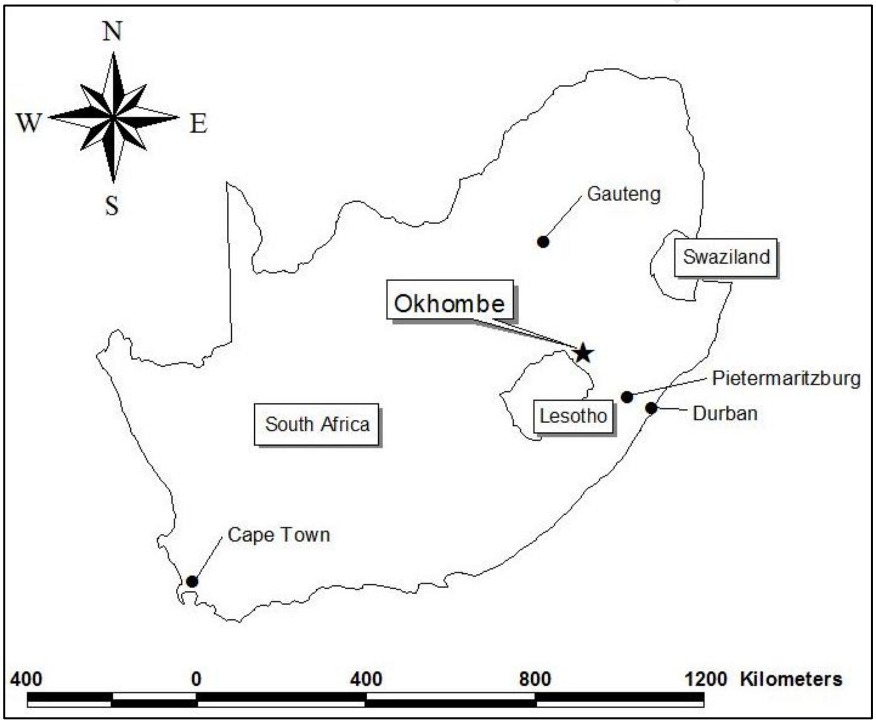

Fig 1: Location of study site, the ward of Okhombe, near Lesotho.

Okhombe has a typically sub-humid climate, and receives the majority of its rainfall during summer from October until March (Schulze 1997), with between 800 and 1,265 mm per annum (Camp 1999). Despite high rainfall, Okhombe is subject to occasional drought (Everson et al. 2007). Temperatures change considerably with season; summers are moderate, while winters are cold, with mean temperature between 11.5°C and 16°C. Frost is common through the coldest months of June and July, and although snow falls in the higher reaches of the Drakensberg, it seldom reaches Okhombe.

The valley of Okhombe lies within the Montane Belt climax plant community, which covers an altitudinal gradient from 1,250-1,800 m, mostly consisting of river valleys and their slopes (Killick 1963). The vegetation of the area is classified as Highland Sourveld (Acocks 1953), or Northern Drakensberg Highland Grassland (Mucina & Rutherford 2006) and is predominantly forb-rich sourveld grassland with small patches of montane forest and shrubs typically found in fire refugia such as steep kloofs. Frequent burning regimes are the dominant influence on the vegetation, referred to as fire climax grassveld (Tainton 1999). In areas utilised by local communities, the grassland community is secondary, largely due to increased burn frequency (Everson et al. 2007). Okhombe is part of the Amazizi Traditional Authority of Upper Thukela, and is comprised of six subwards; Mpameni, Ngubela, Oqolweni, Mahlabathini, Enhlanokhombe and Sgodiphola.

Okhombe was one of the wards which was subjected to the controversial "betterment planning" of the past regime. Betterment planning was the restructuring of communal settlements, in order to facilitate the supply of utilities and improve accessibility. This included the forced resettlement of communities from a widely dispersed pattern to more densely clustered sub-wards or villages. However, the betterment system also implemented a formal cattle management plan to increase productivity and maintain veld quality. Through this initiative, fences were erected to exclude the cattle from the lowlands during the crop growing season. Although the betterment system was abandoned following the collapse of the apartheid government in 1994, the community has since resurrected the grazing fences. With this system in place, communities are better able to manage cropping and grazing. Typically, cattle graze the hillslopes in summer, and then are returned to the valley in winter to graze the remains of crops, as the grasses have by then become largely unpalatable (Everson et al. 2007). The vegetation of the area is also regularly burned in late winter and early spring to promote grass shoot growth, although burns frequently occur out of season.

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2.3.3.2. Design and treatments

The study encompasses four sites within communal rangelands at Okhombe. At each site, an area has been fenced to exclude grazing, and the OMG have carried out extensive restoration, using a range of interventions outlined earlier. In this study, the fenced exclosure at each site comprised the restored treatments or plots used in this study, while a directly adjacent, open plot of similar aspect was chosen for each degraded treatment. Each of the four sites is located within a different sub-ward of the valley of Okhombe, namely; Mpameni, Ngubela, Mahlabathini and Oqolweni. In addition to these four paired sites, a site of relatively good condition within National Park boundaries was also assessed to provide a comparison (hereafter referred to as the "pristine site").

2.3.3.3. Vegetation cover and species diversity

The Levy-bridge point intercept method (Levy & Madden 1933) was used to identify plant species at 200 points per site. A levy bridge consist of a platform which holds ten metal spokes upright at an even distance across a meter span, marking ten points when placed on the ground. The Levy-bridge was placed parallel to contour lines, and located randomly within 5 m either side of the two transect lines. The two transects where located by choosing a random starting point 1m from the base of the plot and running the transect perpendicular to contour lines up the slope. If no plant was encountered at a given point, then the nearest neighbour plant was chosen. These points were used to calculate richness, diversity and veld condition (see below). The distance diameter method for grass tufts (Hardy & Tainton 2007) was used to measure basal cover at 50 points per site within the same area. A point was taken at every two meters along the same transect lines that were used for LFA, and then the distance to the nearest grass tuft or large non-perennial forb was recorded, as well as its diameter (measured perpendicular to the transect line). Species diversity was calculated for each site using the Shannon-Weiner diversity index, H. This was calculated using the following equation, where Pi is the proportion of each species in the sample:

$$H = - \sum Pi(\ln Pi)$$

2.3.3.4. Veld Condition and grazing capacity

The ecological index method for veld condition assessments (Tainton 1999) was used to compare restored, degraded and the pristine plot with a recognised benchmark site for the vegetation of the area, namely Bioresource Group 8 (Camp 1999). Species identified at 200 points along the transects within each plot were classified into different ecological categories (Increaser I, II, III, or Decreaser species) and assigned a relative grazing value. Ecological classification is determined by the species response to intensity and frequency of defoliation, allowing for specific groupings to be associated with different disturbance regimes. Decreasers grasses are numerous in good veld, and decrease in number when the veld is overgrazed or undergrazed, and include the highly palatable *Themeda triandra*. Increaser I species are abundant in underutilised veld, and are typically unpalatable, robust climax species. Increaser II species increase in overgrazed veld, and are typically pioneers or subclimax species such as *Aristida adscensionis*. Increaser III species are typically common in overgrazed veld and are strong competitors during overgrazing, but suffocate quickly when veld becomes underutilised. After species where classified each plot was then given a total score, based on the grazing values and abundances of the species found at the site. The total plot score was then expressed as a percentage of the recognized benchmark score from Camp (1999). Grazing capacity was then calculated using an equation for current grazing capacity (Hurt 1989) based on veld condition score, plant composition, slope and soil erodibility factor. The formula applied was as follows;

$$CGC = PGC \times CF + TF + SEF$$

Where:
CGC = Current Grazing Capacity
PGC = Potential Grazing Capacity of the recognized benchmark site for the area.
CF = (Composition score + the number of units of palatable increaser I species in excess of the benchmark)/100.
TF = Topographic Factor, where sites with a slope from 0°-15° = 0.2 and steeper slopes = 0.0.
SEF = 0.055F – 0.005. Where F is the erodibility rating of the soil series of the site, as determined subjectively by professional opinion, and scored below (Table 1). Due to high erodbility, SEF = 0.11 at all sites except for the pristine site, where SEF = 0.33.

2.3.3.5. Hydrology

A common technique for measuring hydraulic conductivity of soils is the double ring infiltrometer (Bower, 1986, Perroax & White 1998). This technique is performed *in situ* at the soil surface, and provides a relatively uncomplicated and rapid means of measuring the rate at which water can infiltrate soils. The double ring infiltrometer operates under a ’ponded’ situation, where the soil surface is completely saturated. Saturated infiltration rates

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(i.e. double ring) can often be orders of magnitude greater than unsaturated, due to the presence of macropores and small fissures in the soil surface (Colloff et al. 2010; Youngs 2000; Dirksen 2000). Macropores are the tunnels and interstitial spaces created by endogeaic biota or soil processes and create an easy path for water to penetrate down the soil profile. Numerical modelling has shown that falling and constant head conditions yield very similar results on fine soils (Wu et al. 1997), such as those in Okhombe. Although the double ring technique was designed for non-sloping, level terrain, many studies have utilised it on slopes of up to 30% (Watson & Luxmoore 1986; Elliot & Efetha 1999; Joel & Messing 2000). Past research on hillslopes of up to 20% have demonstrated no impact of slope on measurement (Bodhinayake, 2004), and consequently slope was not considered to be an issue at any of the four study sites. For this study, saturated infiltration under ponded conditions was measured using a custom, mini double ring infiltrometer with an inner diameter of 8 cm. Due to the long period of time required for infiltration to reach steady state on fine clay soils, ten double ring infiltration replicates were performed per plot. Steady state is only achieved once the upper soil profile is saturated, and is typically much lower than the first few infiltrations which are rapidly absorbed by the dry soil. To ensure accuracy, measurements were taken at regular intervals (How far the water level has fallen in the tension infiltrometer every 60 seconds) at each of the 10 plots until the rate of infiltration became steady, for a maximum of 60 minutes, after which the mean of the last three infiltration readings was calculated.

2.3.3.6. Soil measurements

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Ten soil samples were collected at each site, comprising of a standard 10 cm diameter ring for bulk density, and then a further ±1 kg of soil in a swath surrounding the ring to a depth of 5 cm. The bulk density cores were oven dried at 50 °C for two weeks, until mass remained constant, and then weighed in order to determine average bulk density per site. Bulk density was adjusted for stone content by removing the stones, weighing them and then applying a correction factor appropriate to the underlying bedrock.

50g of dry soil from each soil sample was analysed for total carbon and nitrogen content in order to determine carbon storage and nutrient cycling. The Walkley-Black (1934) modification of the chromic acid titration method was used to determine organic carbon content, while nitrogen content was analysed using LECO apparatus. Both carbon and nitrogen analyses were performed at BEMLABS, Strand, Western Cape.

2.3.3.7. Landscape Function Analysis

The full Landscape Function Analysis (Tongway & Hindley 2004) technique was conducted along two representative 50 m transects within each treatment. Transect locations were recorded with a GPS and the bearing of each transect was recorded along with the general hillslope aspect using a magnetic, lensatic engineer compass. The LFA technique consists of classifying the length of the 50m transect in either "patches" or "interpatches" – based on where nutrients may be trapped by any obstructions, such as grass clumps or fallen branches. The proportion of patch to interpatch length forms the Organisation Index. Different types of interpatches and patches are then further divided into distinct query zones, and a series of 11 qualitative tests is conducted at 5 points for each query zone along the transect. The variables measured are: rainsplash protection, vegetation cover, litter, cryptogam cover, crust brokenness, soil erosion type & severity, presence of deposited materials, soil surface roughness, surface nature, slake test, and soil texture. These variables comprise the soil surface assessment which is used to calculate the three indices of rangeland health, namely; Soil stability, infiltration and nutrient cycling. Landscape Function Analysis indices (Organisation index, stability, infiltration, nutrient cycling) were generated in Microsoft Excel using a standard macro template (Tongway & Hindley 2004), and range from 0-100.

2.3.3.8. Statistical Analysis

IBM SPSS Statistics 19 was used for all analyses, except for the Lords Range test, which was performed manually in Microsoft Excel 2007. All quantitative data were tested for normality by performing both Kolmogorov-Smirnov (with Lilliefors correction), and Shapiro-Wilks tests, and also by examining normal probability plots (Q-Q plots). When the data were found to be normally distributed, an ANOVA was performed to test for significant differences between groups. Homogeneity of variance was examined using Levene's test, and if heteroscedascity was found, ANOVA results were confirmed with both Welch and Brown-Forsythe robust tests for equality of means. If variances were equal, a post-hoc Tukey HSD test was then performed in order to determine what sites differed significantly within groups. However, if variances were unequal, a Games-Howell post hoc test was used instead. If data was not normally distributed, a non-parametric Kruskal-Wallis test with multiple pair-wise

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comparisons was performed using the Bonferroni correction to find any significant differences between and within groups. The qualitative LFA results were tested for significant differences using Lord's Range test for small sample sizes (Lord, 1947). Correlations were performed using the non-parametric Spearman's rank correlation co-efficient.

2.4. Results

For simplicity and ease of presentation, site names are abbreviated as follows:

Table 1: Abbreviations of site names used in graphs.

Abbreviation	Site Name
MHBDGD:	Mahlabathini Degraded
MHBRHB:	Mahlabathini Restored
MPMDGD:	Mpameni Degraded
MPMRHB:	Mpameni Restored
NGBDGD:	Ngubela Degraded
NBGRHB:	Ngubela Restored
OKWDGD:	Oqolweni Degraded
OKWRHB:	Oqolweni Restored
RNNP:	Royal Natal National Park
BRG8	Bioresource Group 8 benchmark

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2.4.1. Vegetative Composition

Species Richness

No clear pattern was seen in terms of species richness between restored and degraded plots (Table 2). Species richness was greater on restored than degraded plots at Ngubela and Oqolweni, but lower on the restored plot at Mahlabathini, and identical at both plots at Mpameni (Table 2).

Species Diversity

Species diversity also showed no clear patterns (Table 2). Species diversity was greater on the degraded plots at Mahlabathini and Oqolweni, but lower for Mpameni and Ngubela (Table 2).

Basal Cover

Basal cover was lower on restored than degraded plots for all sites except Mahlabathini. Basal Cover at Royal Natal National Park was low compared to many of the other sites within the communal rangelands (Table 2).

Veld Condition Assessment

Veld condition assessments showed mixed results, with only the restored plot at Mahlabathini showing a noticeable increase over degraded sites (Table 2). All plots within the communal rangeland and at Royal Natal National park were substantially lower than the benchmark site (Table 2).

Grazing Capacity

Grazing capacity was uniformly poor at all communal areas, with the exception of the Mahlabathini restored site (Table 2). This site was comparable in grazing capacity to the Royal Natal National Park site, although both were still substantially lower than the Bioresource Group 8 benchmark (Table 2).

Table 2: Indicators of vegetative composition per plot at each site.

Site	Species Richness	Species Diversity	Basal Cover	Veld Condition (%)	Grazing Capacity (Au/ha)
MHBDGD	31	2.6	15.3	25	0.13
MHBRHB	19	1.6	21.3	36	0.28
MPMDGD	16	2.0	14.2	21	0.12
MPMRHB	16	2.1	12.9	21	0.12
NGBDGD	22	1.7	22.8	30	0.14
NGBRHB	31	2.8	14.0	25	0.13
OKWDGD	15	1.8	28.6	35	0.15
OKWRHB	17	1.7	19.9	30	0.14
RNNP	33	3.1	14.6	39	0.30
BRG8	-	-	12	100	1.80

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2.4.2. Hydrology

Infiltration

Infiltration was found to differ significantly between sites (Fig 2, $H(6)=20.2$, $p < 0.05$). However, although average infiltration rates were consistently higher on restored plots than on degraded (Fig 2), these differences were not statistically significant. Multiple pairwise comparisons revealed that the only plot within the communal area that differed significantly from that at Royal Natal National Park was Oqolweni degraded ($M=-29.1$, $p < 0.05$).

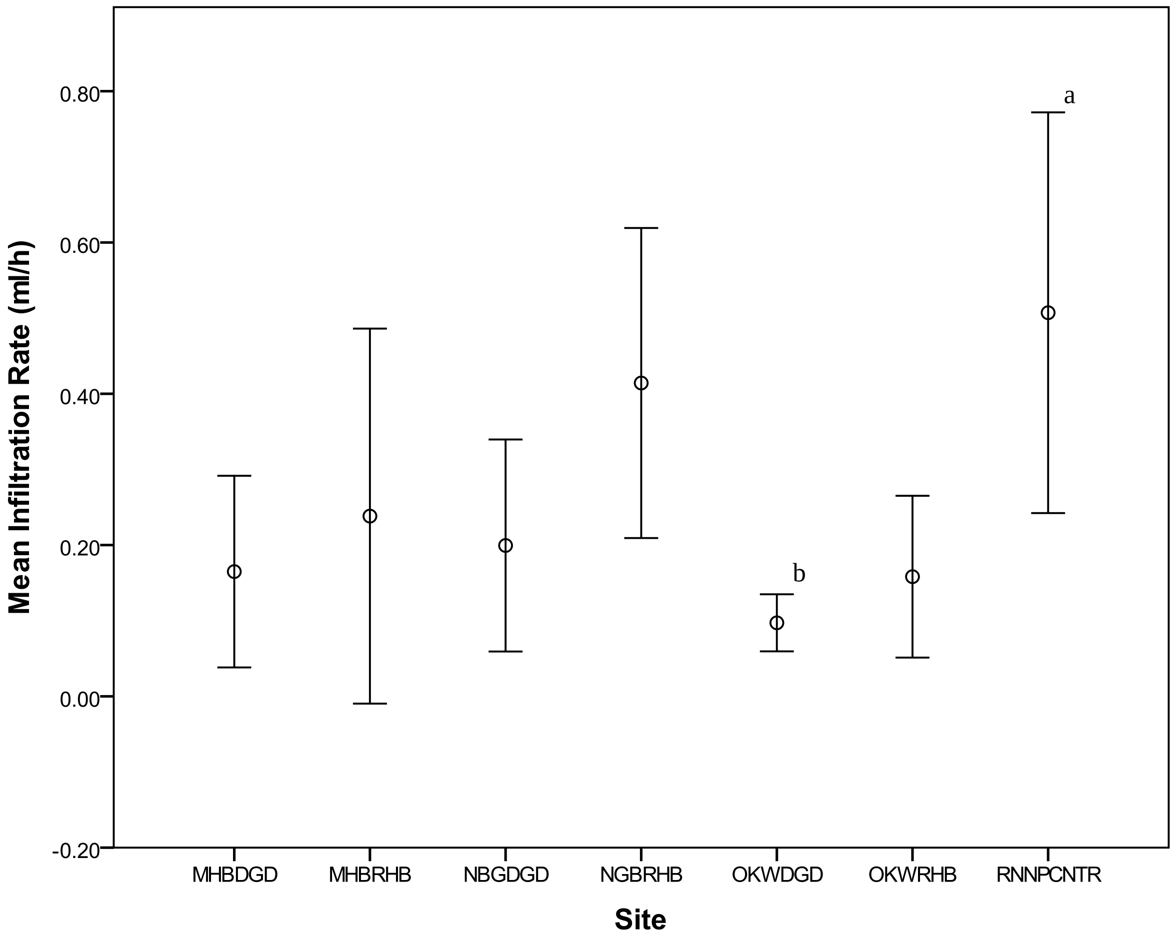

Fig 2: Mean infiltration rates per plot at three sites within Okhombe, and at Royal Natal National Park. Error bars indicate 95% confidence intervals.

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2.4.3. Soil Quality

Carbon Content

Average carbon content was lower on restored than degraded plots for all sites (Fig 3). Carbon content was found to differ significantly between sites ($F(8)=31.80$, $p<0.001$). A Games-Howell post-hoc multiple comparisons test showed that restored and degraded plots differed significantly from each other only at Oqolweni (Fig 3, c-d) where the rehab site was lower, ($p < 0.05$), and that the carbon content at Royal Natal National Park was significantly higher when compared to all other plots (Fig 3, a-b), ($p < 0.001$).

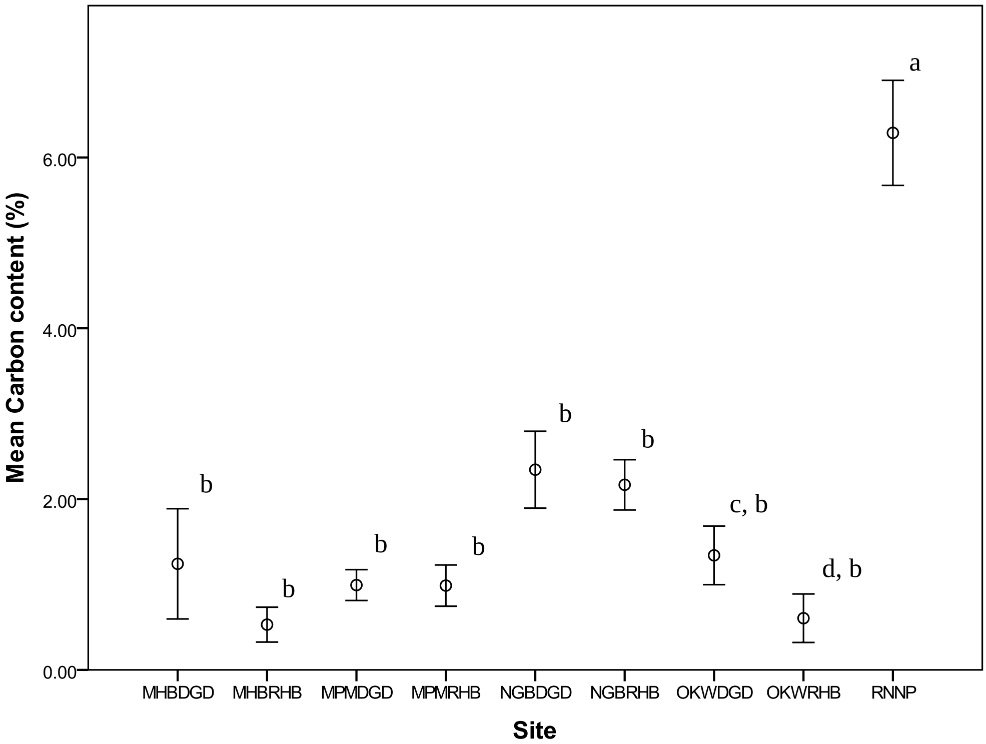

Fig 3: Mean carbon content (%) per plot at all sites, error bars indicated 95% confidence intervals.

Nitrogen Content

Average nitrogen content was lower on restored than degraded plots for all sites except Mpameni (Fig 4). A Kruskal-Wallis test for independent samples found significant differences between groups ($H(8)=65.4$, $p<0.001$). However differences between restored and degraded plots at each site were not significant. Pairwise comparisons performed using the Bonferroni correction revealed that mean nitrogen content at all plots except for Oqolweni & Ngubela degraded and Ngubela restored differed significantly from Royal Natal National Park (Fig 4, a-b) ($p < 0.001$).

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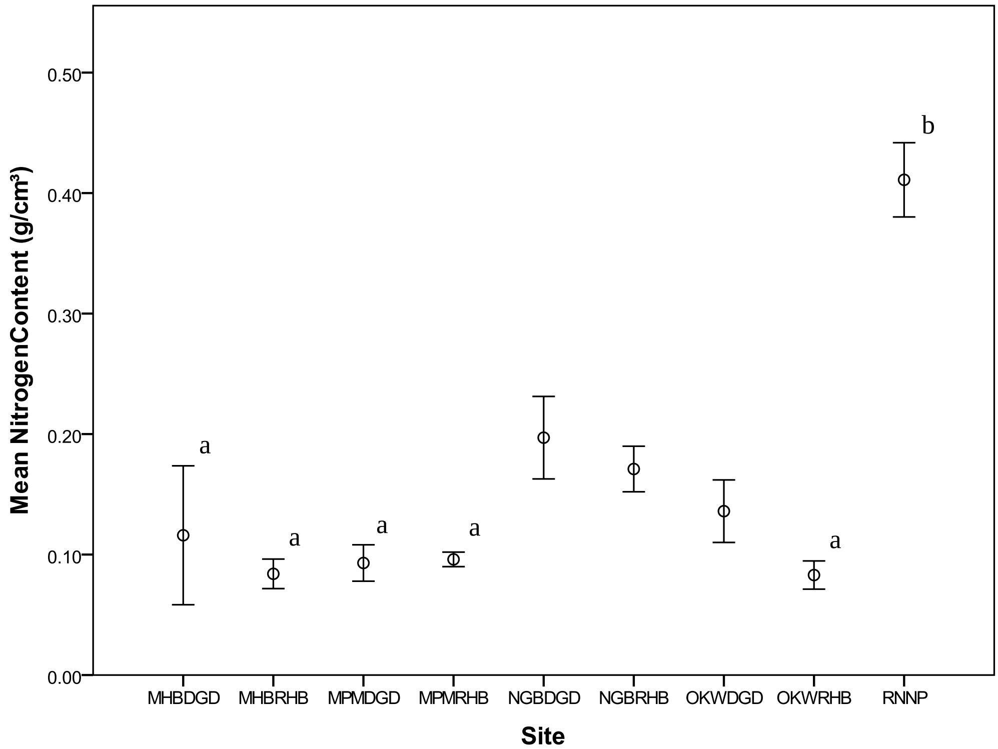

Fig 4: Mean nitrogen content (%) per plot at all sites, error bars indicated 95% confidence intervals.

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Bulk Density

Within sites, density differed significantly only at Ngubela, where it was higher on the degraded plot than the restored plot (Fig 5, $p < 0.05$). Bulk density at Royal Natal National Park was significantly lower than any of the other plots (Fig 5, $p < 0.001$).

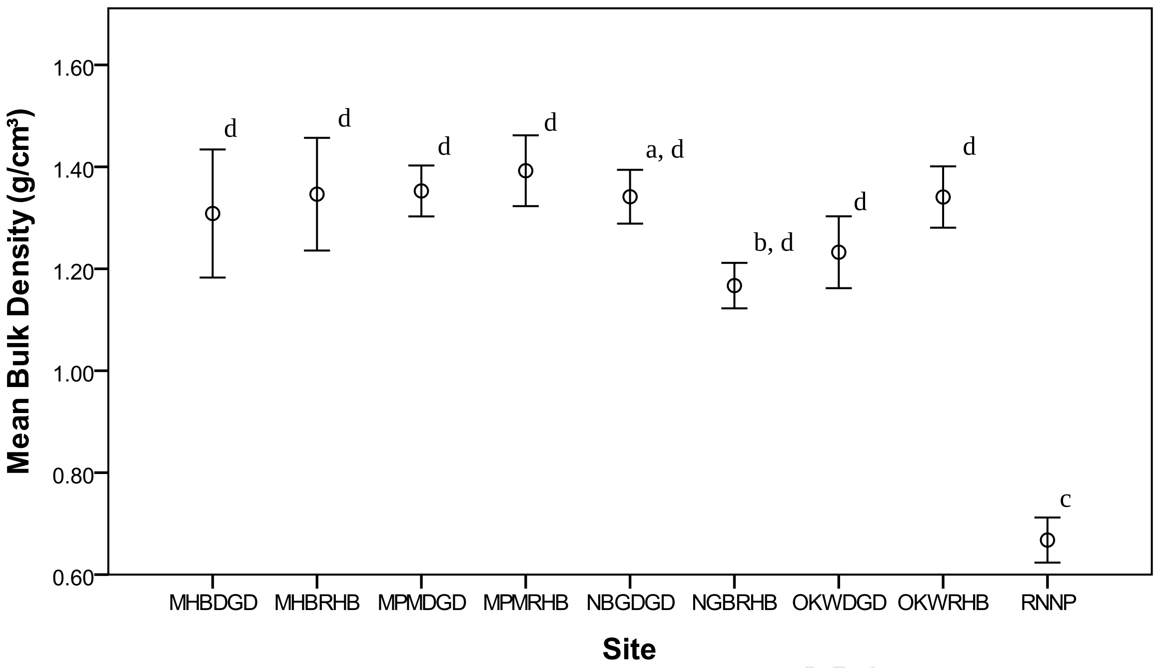

Fig 5: Mean bulk density per plot at all sites, error bars indicated 95% confidence intervals.

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2.4.4. Landscape Function Analysis

Stability Indices

The average stability indices derived from Landscape Function Analysis showed no consistent pattern between restored and degraded sites (Fig 6). Restored sites showed greater stability than degraded sites at Mahlabathini (65.3 vs 62.6) and Mpameni (58.8 vs 56.8), while the opposite was true for Ngubela (58.9 vs 64.5) and Oqolweni (54.6 vs 60.0). None of these differences were statistically significant. Lord's Range test for small sample sizes did find that stability indices at Oqolweni degraded, Ngubela degraded and Mahlabathini Rehab differed significantly from Royal Natal National Park (Fig 6, n=2, L > 1.71).

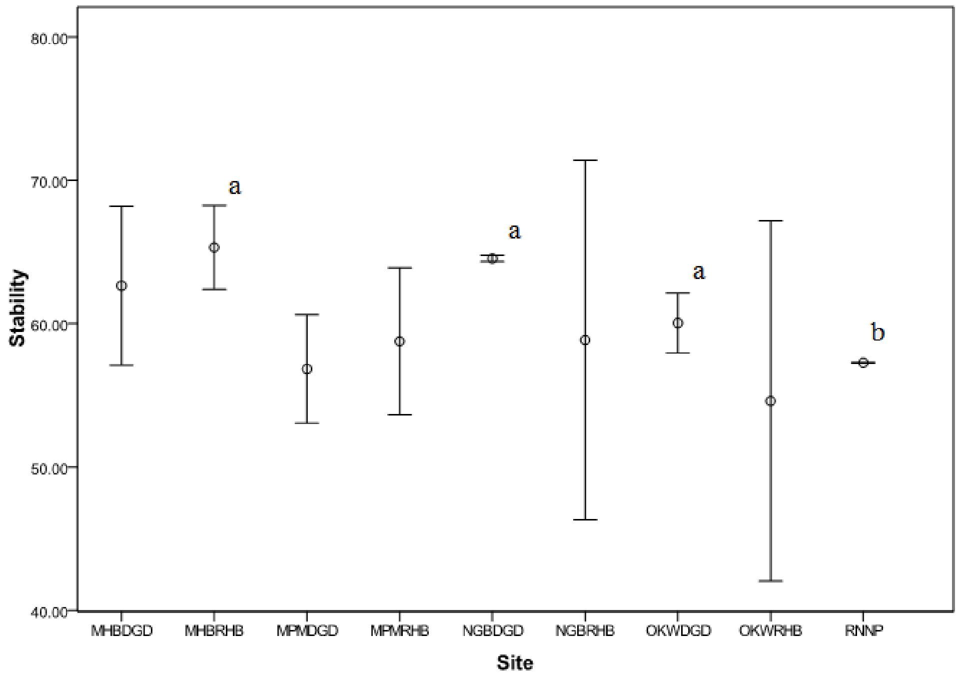

Fig 6: LFA calculated Stability indices per plot at all sites, error bars indicated 2 standard deviations from the mean, as calculated by the LFA macro (Tongway & Hindley, 2004)

Infiltration Indices

Infiltration indices showed a similar trend to the stability indices (Fig 7). Restored sites showed greater infiltration than degraded sites at Mahlabathini (33.2 vs 31.3) and Mpameni (26.9 vs 26.7). However, the opposite was true for Ngubela (28.2 vs 29.8) and (25.6 vs 28.7). Again, Lord's Range test for small sample sizes found none of these differences to be significant (n=2, L < 1.71).

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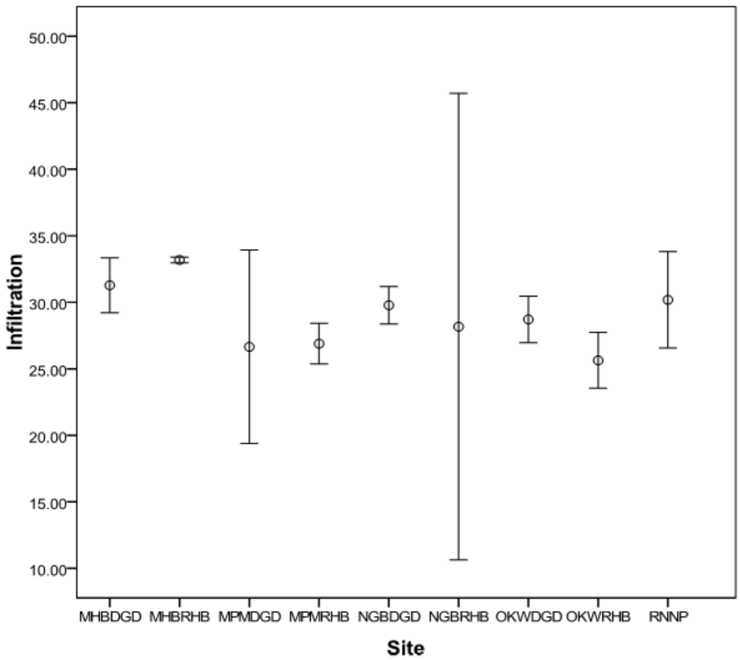

Fig 7: LFA calculated infiltration indices per plot at all sites, error bars indicated 2 standard deviations from the mean, as calculated by the LFA macro (Tongway & Hindley, 2004)

Nutrient Cycling Indices

Average nutrient cycling indices were lower for restored than degraded treatments at all sites except for Mahlabathini (Fig 8). However, Lord's Range test for small sample sizes again found none of these differences to be significant (n=2, L > 1.71). The site at Royal Natal National Park had the lowest nutrient cycling index of any site (Fig 8, 18.5).

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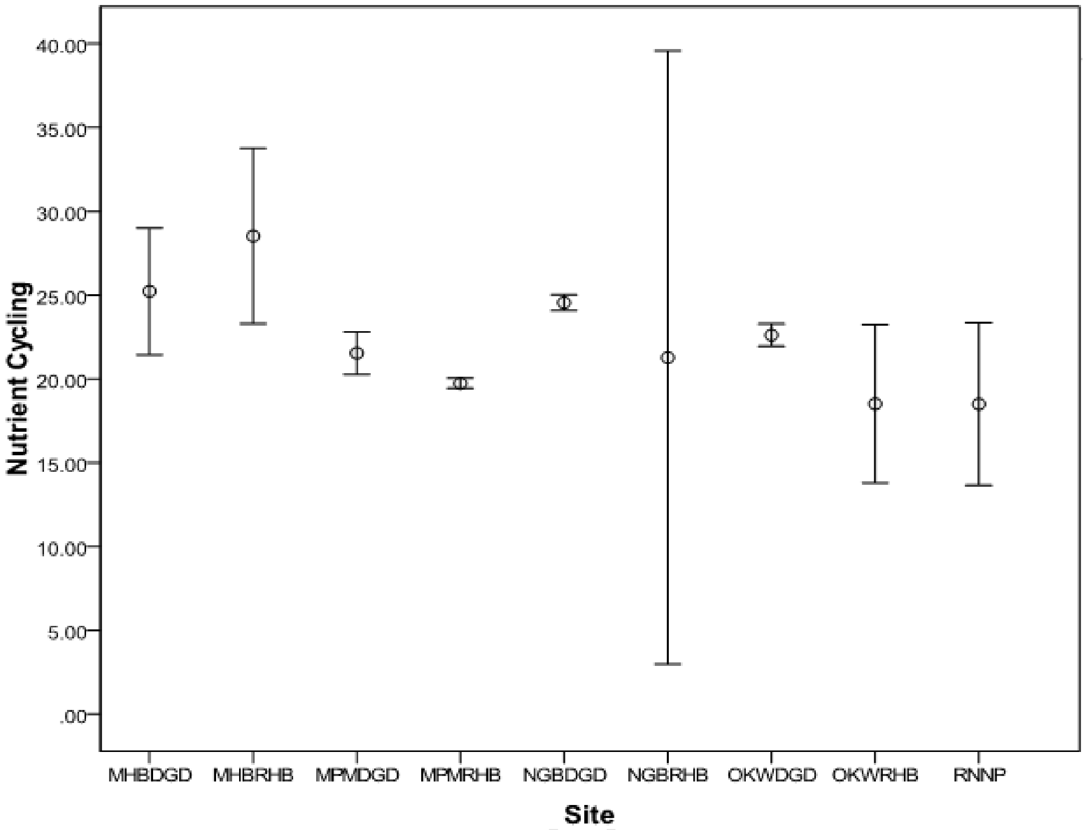

Fig 8: LFA calculated Nutrient Cycling indices per plot at all sites, error bars indicated 2 standard deviations from the mean, as calculated by the LFA macro (Tongway & Hindley, 2004)

Organisation Index

Average landscape organisation indices were lower for restored than degraded treatments at all sites except for Mahlabathini (Fig 9). Again however, Lord's Range test for small sample sizes found none of these differences to be significant (n=2, L < 1.71). The organisation index of Royal Natal National Park was significantly lower than any of the other sites (n = 2, L > 1.71) except for Oqolweni restored – which had a far greater range than the others (Fig 9).

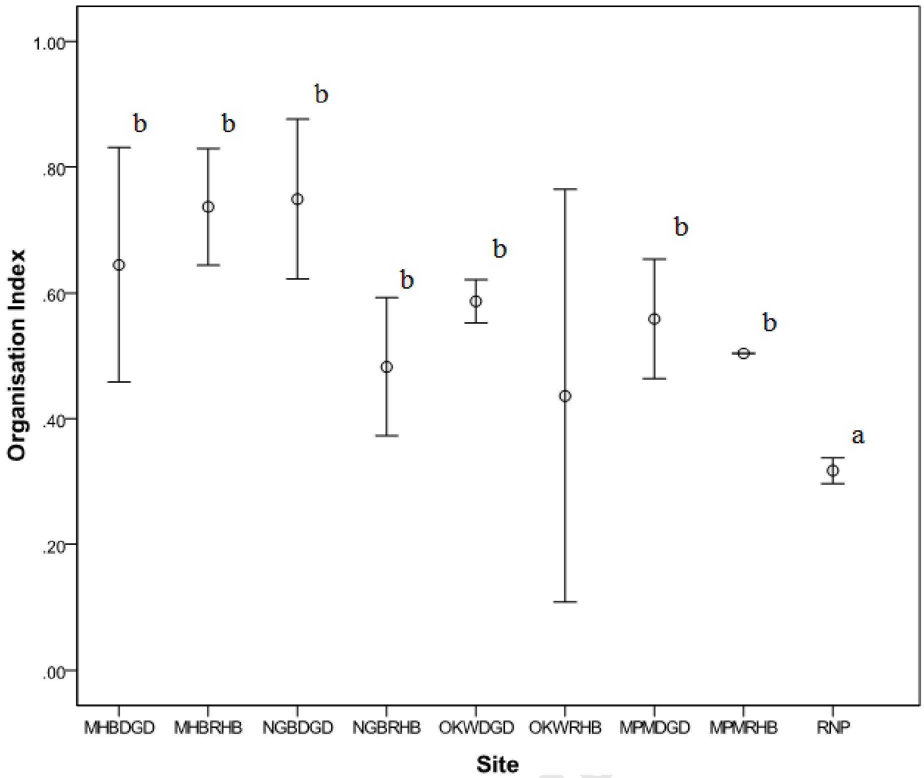

Fig 9: LFA calculated Organisation indices per plot at all sites, error bars indicated 2 standard deviations from the mean, as calculated by the LFA macro (Tongway & Hindley, 2004)

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2.4.5. Correlations between LFA and qualitative data

Infiltration

The relationship between qualitative infiltration and the LFA derived infiltration index was investigated using Spearman-rank correlation coefficient (Fig 10). No significant correlation was found between the two variables (R=0.214, n = 7, p > 0.05).

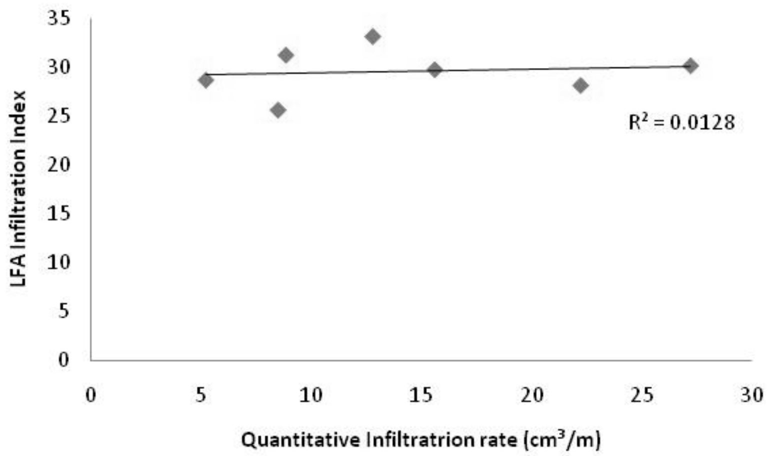

Fig 10: Correlation between quantitative infiltration rate and qualitatively assessed LFA infiltration index.

Soil stability

The relationship between qualitative carbon content and the LFA derived soil stability index was investigated using Spearman-rank correlation coefficient (Fig 11). No significant correlation was found between the two variables ($R = 0.05$, $n = 9$, $p > 0.05$).

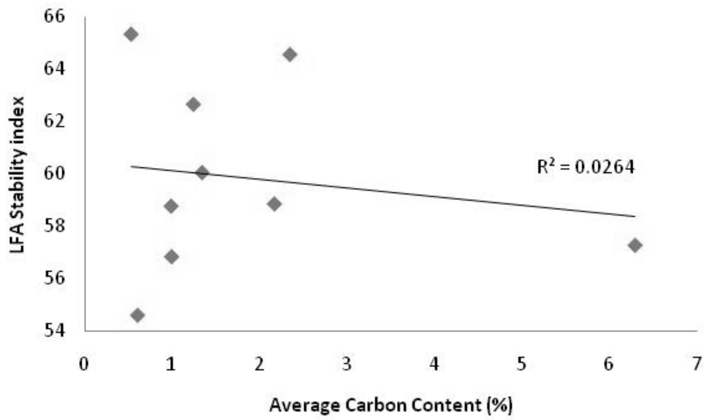

Fig 11: Correlation between soil carbon content and qualitatively assessed LFA stability index.

Nutrient Cycling

The relationship between average carbon – nitrogen ratio and the LFA derived Nutrient Cycling index was investigated using Spearman-rank correlation coefficient (Fig 12). No significant correlation was found between the two variables ($R = -0.32$, $n = 9$, $p > 0.05$).

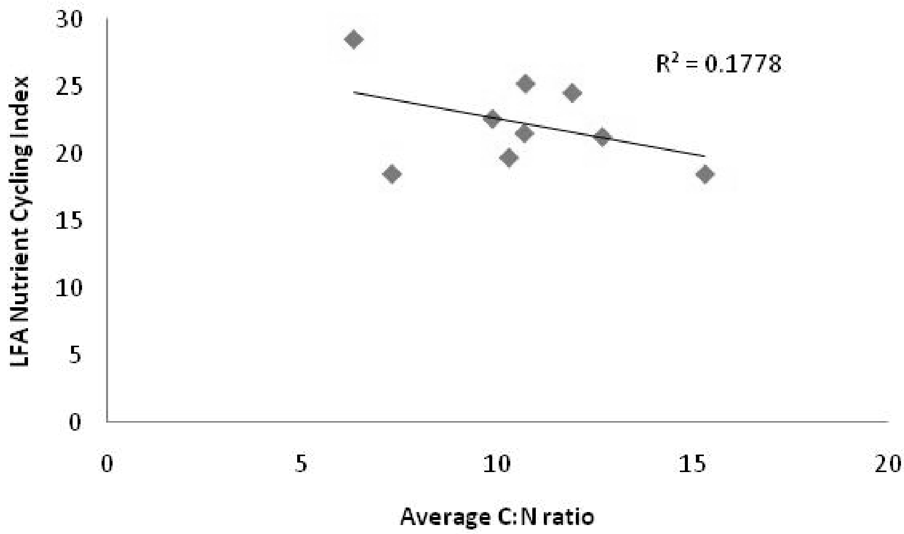

Fig 12: Correlation between average C:N ration and qualitatively assessed LFA Nutrient Cycling index.

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2.5. Discussion

2.5.1. The Impacts of Restoration on Vegetation

Basal Cover

Vegetation cover is well recognised as an effective and useful indicator in measuring the direction of plant succession, and is one of the most frequently used variables when evaluating restoration (Ruiz-Jaen & Aide 2005). The primary goal of many restoration projects is the rapid recovery of vegetation cover, as this stabilises the soil surface, provides rainsplash protection and reduces erosion (Armstrong & Mitchell 1988). Furthermore, vegetation cover is key to many of the processes which ultimately drive a more full recovery (Tongway & Hindley 2004; Perrow & Davy 2002; Toth et al. 1995; Young 2000). Previous studies have found that low productivity sites where vegetation cover is sparse and slow to recover are more prone to erosion than well vegetated, highly productive sites (Ninot et al. 2008), reinforcing the importance of near-pristine basal cover.

Basal cover at all sites except for Mahlabathini was lower on restored than adjacent openly grazed plots (Table 2), which is seemingly counterintuitive in light of reduced grazing pressure. However, on degraded sites, increased basal cover is due largely to the invasion of *Paspalum notatum*, an exotic grass. *Paspalum notatum* thrived on heavily grazed sites such as Ngubela and Oqolweni and previous studies have found it to be tolerant of even the most severe defoliation (Pakiding & Hirata 2001). It is likely then, that *P. notatum* has a strong competitive advantage on heavily grazed sites. Further evidence that basal cover is artificially high on sites dominated by *P. notatum* can be seen in the comparatively low basal cover at Royal Natal National Park and the bioresource group 8 benchmark, which are reflective of a situation closer to a pristine state (Table 2). In this instance, basal cover still provides an important indicator, however rather than simply seeing a unidirectional increase as a positive change, it must be recognised that greatly increased basal cover may represent a deviation from a normal state, which may not necessarily be desirable.

Compared to most indigenous tufted species, *P. notatum* forms a dense mat of tough, grazing resistant stolons. It has been recognized as a useful cover forming species in many restoration projects around the world, where it has been found to help to bind soils, trap nutrients and

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reduce erosion (Hancock et al. 2010; Koo et al. 2005; Kalmbacher and Martin 1998; Reynolds et al. 1999; Xia and Shu 2001 *in* Xia 2004, Xia 2004). In the restored plot at Mahlabathini, where the swards of *P. notatum* are no longer grazed, the grass had begun to accumulate a layer of organic matter below the soil surface. This indicates that if grazing pressures are removed, passive restoration may be promoted. In this sense, heavily grazed areas dominated by *P. notatum* can be seen as an alternative stable state for communal rangelands in Okohmbe (Aronson et al. 1993; Muller et al. 1998). Deliberate introductions of *P. notatum* in humid grasslands in America have been considered a success due to its tolerance of low nutrients and heavy grazing (Hoveland 2000). Preliminary trials using *P. notatum* stolons to restore degraded areas have been put in place at Oqolweni, but it is still too early to yield significant results. Whether these large patches dominated by *P. notatum* are able to be replaced by more representative indigenous vegetation once grazing pressure is removed remains to be seen. If this is not the case, then their value as a replacement ecosystem must be determined.

Species Richness and diversity

Neither species richness nor diversity displayed any noticeable trends between degraded and restored sites (Table 2), although all sites within Okhombe were lower in both species diversity and richness compared to the reference site at Royal Natal National Park. Only Ngubela showed a substantial increase in species diversity and richness at the restored site. As Ngubela is the oldest of the restored plots, this suggests that a long period of exclusion is required in order to observe an increase in improvement in species composition. Long recovery times are expected in these sourveld regions, due to low dispersal and germination success of palatable grasses such as *Themeda triandra* (Everson et al. 2009). Consequently it may take a particularly long time before the palatable fire climax species return, and active planting may be necessary.

Species diversity was also high (H'=2.6) on the Mahlabathini degraded plot. Grazing has been shown to increase evenness in some systems by reducing more abundant species (Hickman & Walter 1996), a process which may be increasing the diversity indices on degraded sites. However, this in itself is not sufficient to explain the high diversity seen on Mahlabathini's degraded plot, as species richness was also comparatively high (Table 2). This may indicate that it has experienced some protection from grazing, possibly due to local

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topographic features, as grazing would likely decrease species richness by removing palatable species. Although frequently used as an indicator of overall diversity, plant species diversity may not necessarily correlate with other indices of faunal diversity, such as insects or birds (Mortimer et al. 1998; Bakker & Berendse 1999; WallisDeVries et al. 2002). Accordingly, further investigation would be required in order to determine a more complete view of the difference in species diversity between degraded and restored plots. Restoration does not always improve species diversity, and species richness has even been found to decrease with age since restoration (Sluis 2002). This highlights the importance of continued management, particularly in the Drakensberg grasslands, where many species are highly adapted to regular disturbance (Everson & Tainton 1984).

Veld Condition Assessments and grazing capacity

Veld condition is a measure of the health of veld in terms of its successional stage, susceptibility to erosion, and its ability to provide forage for stock (Trollope et al. 1990). Veld condition not only provides an assessment of rangeland health, it also indicates predominant functional grass types, and the degree and type of grazing pressure being exerted on the system (Tainton, 1999). Overall, both veld condition and grazing capacity was similar between degraded and restored sites (Table 2), which were again all consistently lower than that at Royal Natal National Park. The only appreciable difference between plots was at Mahlabathini, where both veld condition and grazing capacity was higher under restoration (Table 2). Increaser IIc species (pioneer species which increase in abundance with sever overgrazing) were 20% more abundant on the degraded plot at Mahlabathini, which is indicative of severe overgrazing, particularly when surface soil is lost (Tainton, 1999). The increase in both species diversity and proportion of Increaser IIc species at Mahlabathini may be explained by patches of severely overgrazed, eroded veld which occurred on the more accessible lower slopes of the plot. Due to the steep topography of the site, it is likely that only the more exposed fringes of the plot experienced heavy grazing pressure, with less accessible areas retaining a healthier compliment of species. The uniformly poor veld condition and grazing capacity observed at most sites within the communal rangeland suggests that current management practices are either not effective, or that more time is needed before changes will begin to manifest themselves. Reintroducing and adhering to a biennial spring burn regime, coupled with appropriate periods of rest should lead to an improvement in veld condition and grazing capacity (Everson & Tainton, 1984). This is because a biennial spring burn regime allows enough time for the growth of reproductive

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tillers in plants such as *Themeda triandra*, and avoids damaging plants during the important summer growth season which would hamper reproductive output (Everson et al 2009). Adjusting the burn regime will need to be coupled with similar periods of rest from grazing pressure in order to be successful. However, as previously mentioned, even given such measures highly palatable species may only return when physically reintroduced due to extremely limited dispersal in the sourveld (Everson et al. 2009).

2.5.2. The Impacts of Restoration on Hydrology

Infiltration

The recovery of healthy soil infiltration is critical to restoration success, as infiltration is a primary mechanism through which soil water reserves are replenished, and plant growth enhanced (Ritchie 1998). Although differences were not statistically significant, a clear trend was present where average infiltration rates were consistently higher on restored than degraded plots (Fig. 2). This suggests that infiltration has improved in response to restoration. As underlying abiotic factors were likely similar between directly adjacent plots, the increased infiltration rates observed on the restored site are likely due to biotic factors. Such factors may include the recovery of natural soil processes and endogeic microfauna as a result of relief from trampling, as these have been shown to increase infiltration through the creation of macropores in the soil (Beven & Germann 1982). Increased infiltration also results in greater base flows in the dry season, which is an important water resource for the local community.

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It is important to note that although basal cover was greater on almost all degraded sites, this did not translate into increased infiltration rates. This is likely because the tough, stoliniferous roots of the alien grass *P. notatum* are typically tightly packed, and were often found growing on physically crusted soils. On slopes, decreased infiltration typically results in increased run-off. Increased run-off on degraded sites may lead to enhanced erosion, as stronger surface flows possess greater erosive potential, and can also wash nutrients and organic matter off site (Schlesinger et al. 1999, Schlesinger et al. 2000). The oldest site (Ngubela) showed a greater difference in infiltration rates between plots than the most recently restored site (Oqolweni), potentially indicating that infiltration rates are improving with age since restoration.

Infiltration rates were also consistently more variable on restored sites (Fig. 2). Even a single macropore can greatly increase saturated infiltratrion rate, and due to being randomly distributed across the soil surface (Beven & Germann 1982), their presence would lead to an increase in variation of saturated infiltration rates. Thus the increased variation in infiltratrion rates me be due to increased bioturbation (Beven & Germann 1982). If this is the mechanism behind the greater variation, then it implies that restoration is enhancing natural processes, which will in turn further accelerate recovery. However, further research is needed to verify what is driving these differences in variation, although abiotic factors are unlikely responsible due to the close proximity of restored and degraded sites.

The average infiltration rate at the Royal Natal National Park site was considerably greater than any of the plots within Okhombe (Table 2). This was due to the presence of a well developed layer of thick organic mulch, which has been shown to increase infiltration and decrease evaporation (Reicosky et al. 1995). The variation in infiltration rates was also the highest at RNNP, which is again potentially indicative of a high density of macropores and active bioturbation (Beven & Germann 1982).

2.5.3. The Impacts of Restoration on Soils

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Carbon & Nitrogen

Soil carbon and nitrogen content determined through chemical analysis provides a robust measurement of the nutrient status of soils (Schoenholtz et al. 2000). Soil nutrient status is an important ecological indicator which can provide important information on the health of a given ecosystem (Reganold & Palmer, 1995). Organic carbon provides a direct measurement of total soil organic matter, which plays an important role in many processes, including enhancing nutrient cycling, soil aggregate stability and productivity (Mills & Fey 2003). Differences in carbon and nitrogen content between restored and degraded sites were slight. Where differences were noticeable, both carbon and nitrogen content were lower on restored than degraded sites (Fig. 3&4). This may be due to cattle importing nutrients from less degraded areas through manure and urine. This was observed in studies on the Canadian prairie, where increased grazing resulted in greater soil organic carbon due to increased manure deposition (Smoliak et al. 1972). Significantly higher organic carbon on the reference site in Royal Natal National Park was, due to the presence of thick organic mulch on the soil surface (Fig. 3). A similar increase was seen in nitrogen content at the Royal Natal National

Park site, again due to well conserved soils and associated processes (Fig. 4). This suggests that soil restoration is a long term process and that considerable time is still required before sites within the communal area attain a level where ecosystem processes are approaching a more functional state. Soil recovery however, is likely to take a great deal of time to recover from such severe erosion. Unfortunately, no prior baseline data was available for this study for any of the sites. However, as site selection for restoration projects in Okhombe typically focused on severely degraded areas (Terry Everson, *pers comm*), it is likely that most if not all of the restoration plots were initially in considerably poorer condition than adjacent openly grazed rangeland. This would lead to an underestimation of restoration success, and in this sense, the evaluation can be seen as erring on the side of caution. Accordingly, the lower carbon and nitrogen content on restored sites could simply be an artefact of a poorer starting condition.

Bulk Density

Bulk density is a direct measure of soil compaction, which is particularly robust when soils are similar between compared sites (Hakansson & Lipiec 2000). Trampling by livestock leads to predominantly deleterious impacts on rangeland processes, either by compacting soils and reducing infiltration or disturbing the soil surface and rendering it more susceptible to erosion (Dunne et al. 2011). For example, on fine-textured soils in Colorado, bulk density increased with increasing grazing pressure (Van Haveren 1983). However, some studies have also found unexpected responses to trampling (Dunne et al. 2011). For example, at trials performed on rangelands in the Edward Plateu, Texas, no relationship was found between different levels of grazing pressure and bulk density (Thurow et al. 1986).

Bulk density was significantly higher on all plots within the communal rangelands when compared to Royal Natal National Park (Fig. 5). Again, this is due to well-developed A and O soil horizons at this site. Bulk density between plots was only significantly different at Ngubela, where the restored plot showed lower bulk density than the degraded plot (Fig. 5). This may be because Ngubela is one of the earlier restoration sites, and has had sufficient time for processes such as soil formation and bioturbation to reduce compaction (Wilkinson et al. 2009).

2.5.4. Landscape Function Analysis as a tool to measure impacts of restoration

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Landscape Function Analysis is a rangeland assessment tool designed to easily measure qualitative field indicators to assess the biophysical functioning of a site at the landscape scale (Tongway & Hindley 2004). It was originally developed for the monitoring of degradation and subsequent recovery of mine sites in Australia (Tongway & Hindley 2004). Since then, it has been more widely applied, not only across Australia, but also in Africa, the Middle East, Southern Europe and Asia (Tongway & Hindley 2004). As LFA required very little technical equipment, relying instead on subjective assessments which consider a wide range of variables, it was thought that LFA might provide a valuable tool for the OMG through which to assess restoration effects.

No significant correlation was found between any of the LFA-derived indices and their corresponding quantitative measurements (Fig. 10, 11, 12). Furthermore, when examining indices at specific sites, some obvious discrepancies were noted. For example, the stability index at Royal Natal National Park was one of the lowest values at all sites, despite the fact that it is relatively pristine condition, and showed no signs of erosion (Fig. 6). This site also displayed the highest amount of organic soil carbon, which is positively correlated with soil stability (Mills & Fey 2003). Similarly, both the infiltration and nutrient cycling indices were strongly underestimated by LFA at Royal Natal National Park (Fig. 7, 8). This casts serious doubt on the validity of results obtained through Landscape Function Analysis. Furthermore, in practice, the assessments conducted in LFA proved to be extremely time-consuming, particularly on sites with several distinct types of terrain, or 'query zones' (e.g. bare soil, stony soil, thick mulch). As with any subjective assessment, estimated scores are also prone to significant observer bias (Poulton 1975; Hunt et al. 2003; Booth et al. 2006).

These attributes make LFA particularly poorly suited for a community based monitoring programme, as long term data sets would be compromised should the person responsible for monitoring leave the project. Ultimately, although Landscape Function Analysis may provide an effective tool to detect general trends when applied regularly over a period of time (Tongway & Hindley 2004) by a single observer (Poulton 1975), it is poorly suited for once off comparisons. Because LFA indices were not strongly correlated with any quantitative variables, it would be impossible to utilise LFA to demonstrate the delivery of a specific service, and the precise amount thereof. This renders LFA unable to meet the conditionality needed for a true PES programme (Wunder 2005).

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2.5.5. Ecological theory and the selection of indictors of restoration success

The importance of considering both abiotic and biotic factors and their interaction in restoration ecology is well recognised. Several studies have found that manipulating these factors is a crucial first step for ecosystem recovery (Palmer et al. 1997; Suding et al. 2004; Didham et al. 2005). Whisenant (1999) suggested that thresholds in ecosystems are a result of either biotic interactions, or abiotic limitations. The consideration of abiotic limitations, or a particular set of biotic interactions which constrain a given ecosystem fits the state-transition model suggested earlier (Aronson et al. 1993, Westoby et al. 1989, Briske et al. 2005). Hobbs and Harris (2001) then expanded on abiotic and biotic thresholds in restoration, suggesting that the principal form of restoration intervention is the manipulation of abiotic and biotic factors, and those biotic manipulations should be used to overcome biotic thresholds, and that conversely manipulation of abiotic factors should be used to surmount abiotic thresholds.

In light of the importance of abiotic and biotic interactions and thresholds, it is clear that both of these factors must be taken into consideration in order to assess restoration action. This should be applied from the outset of evaluation, and when choosing what indicators to assess, variables representative of both biotic and abiotic processes should be included. The importance of assessing both abiotic and biotic factors was evident in this study when considering extensive swards of an exotic grass, *P. notatum*, which had become established on degraded plots. This species is extremely tolerant of grazing, and could be considered an example of the stress-tolerance life strategy suggested in Grimes triangle (1977). The dominance of *P. notatum* resulted in increased basal cover, a positive indicator of biotic function. However abiotic indicators showed only negative and neutral changes in degraded areas, with decreased infiltration, and no change in soil nutrient status. This may suggest that although some biotic processes have increased, there still exists a critical threshold where abiotic conditions remain poor. If possible, direct monitoring of actual outputs of ecosystem services will likely yield far more accurate results.

2.5.6. Evaluation of variables as indicators for ecosystem service delivery

This study highlighted some of the issues associated with using different indicators to assess the delivery of ecosystem services. The difficulty in determining a small range of measurable

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variables which are indicative of restoration success has been highlighted by the Millenium Ecosystem Assessment (Carptenter et al, 2007). In this study, a wide range of variables were assessed, however even significant changes in the measured indicators were difficult to attribute directly to a specific ecosystem service or process. Furthermore, multiple indicators were necessary to verify if changes in one indicator variable were leading to other ecosystem effects. For example, vegetation cover, one of the most frequently assessed variables in quantifying restoration successes (Ruiz-Jaen & Aide 2005) was strongly influenced by the presence of large swards of exotic *Paspulum notatum* grass (Table 2), which were not found necessarily indicative of any increase in abiotic functions, such as soil nutrient status and infiltration (Fig 2, Fig 3&4).

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2.6. Conclusion and Synthesis

This study utilised a wide range of both abiotic and biotic indicators in order to assess the ecological impacts of community based restoration actions carried out in Okhombe. Overall, both degraded and restored plots within Okhombe where found to be well below the level of seen at Royal Natal National Park. This is likely due to the abiotic constraints on vegetative growth, such as extremely low nitrogen content (Fig 4) a primary obstacle to restoration success (Pywell et al. 2003).

Basal cover was considerably increased by the presence of large swards of an exotic grass, *Paspalum notatum*. This grass is a stress tolerant species (Grime 1977), and was able to dominated heavily grazed patches on the degraded sites. However, its presence was not related to any improvement in abiotic conditions, contrary to previous work which has shown that vegetation and soil properties exert a strong influence upon each other (Eviner & Chapin 2001). This disconnect between vegetation and soils highlights the importance of considering site specific variability, which has been show to thwart restoration efforts if not taken into account (Wassennaar et al. 2007). Anecdotal evidence from *P. notatum* lawns at the restored site at Mahlabathini suggest that removing grazing pressure may allow for the passive restoration of soil quality, thus acting as an ecosystem of substitution (Bédécarrats 1991, Muller et al 1998). However further investigation is needed to determine if current constraints which are preventing these areas of stoliniferous lawns from accumulating soil organic matter and enhancing soil quality can be overcome, and if they can then be succeeded by indigenous climax species. This would be an example of using restoration to overcome a threshold which is currently keeping the system in a stable, undesirable state, to enable a shift to a more desirable state (Aronson et al. 1993). In the study area, vegetation cover did not provide a clear indicator of recovery, not only because the swards of *P. notatum* confounded results, but also because basal cover in these grasslands is typically low, so a linear increase in basal cover is not indicative of recovery. Overall, there was no clear pattern where either restored or degraded sites were closer in terms of basal cover to the RNNP site.

Due to the low levels of productivity, high levels of stress, and poor propagule pressure in communal rangelands in Okhombe, it is unlikely that successful restoration can be achieved with only the use of passive techniques (Prach & Hoobs 2008; Berendse et al, 1992). Instead,

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active restoration such as that carried out by the OMG is required in order to stabilise soils and return healthy ecosystem function to most severely degraded areas throughout in Okhombe. Even with active restoration, if soil quality has been reduced to a state where plant growth is severely limited, as evidenced by uniformly low soil organic carbon nitrogen content, then restoration to a more functional state may be difficult to achieve (Du Preez & Snyman 1993). The low organic carbon content of soils in Okhombe (as compared to that of RNNP) does suggest that there exists high potential to capture atmospheric carbon (Jones & Donelly 2004) once vegetation begins to recover. Species diversity and Veld condition assessments revealed the need for better management of restored areas, particularly the implementation of appropriate burn and rest cycles, and possibly the physical reintroduction of highly palatable indigenous grass species. Neither veld condition, nor species diversity provided a clear indication that restoration had produced positive effects, as results varied from site to site.

Infiltration was consistently higher on restored plots, indicating that restoration action may be successfully restoring biotic processes such as bioturbation, resulting in increased abiotic function. Indicators of soil quality suggested that restored plots may have initially been in worse condition than the surrounding veld. This is likely due to the local community focusing restoration efforts on particularly degraded sites. While this complicates the ability to quantify the degree to which ecosystem services are being enhanced on restored sites, this study provides valuable baseline data which will be of use in future assessments.

Finally, the qualitative rangeland health assessment technique, Landscape Function Analysis (Tongway & Hindley 2004), was tested against conventional quantitative techniques to determine its efficacy in the area, and if it could provide a useful tool through which the Okhombe monitoring group could evaluate restoration impacts and demonstrate the delivery of ecosystem services. LFA derived indices showed no correlation with quantitative indices, and was found to be highly subjective, time consuming technique. LFA was shown to be an unreliable indicator of processes, and poorly suited to the verification of the delivery of ecosystem services and goods.

University of Cape Town

3. References:

Acocks, J.P.H. 1953. Veld types of South Africa. Memoirs of the botanical survey of South Africa 28: pp 1-192.

Alard, D., J.F. Bance, and P.N. Frileux. 1994. Grassland vegetation as an indicator of the main agroecological factors in a rural landscape: consequences for biodiversity and wildlife conservation in central Normandy (France). Journal of Environmental Management 42: pp 91–109.

Allan, J.D. 2004. Landscapes and Riverscapes: The Influence of Land Use on Stream Ecosystems Annual Review of Ecology, Evolution, and Systematics 35: pp 257-28.

Allsopp N. 1999. Effects of grazing and cultivation on soil patterns and processes in the Paulshoek area of Namaqualand. Plant Ecology 142, pp 179–187.

Anderson, D.C., K.T. Harper, and S.R. Rushforth. 1982. Recovery of cryptogamic soil crusts from grazing on Utah winter ranges Journal of Range Management. 35: pp 355–359.

Angulo-Jaramillo, R., Vandervaere, S. Roulier, J. Thony, J. Gaudet, and M. Vauclin. 2000. Field measurement of soil surface hydraulic properties by disc and ring infiltrometers - A review and recent developments. Soil & Tillage Research 55: pp 1-29.

Armstrong C.L., and J.K. Mitchell. 1988. Plant Canopy Characteristics and Processes Which Affect Transformation of Rainfall Properties. Transactions of the American Society of Agricultural and Biological Engineers 31 (5): pp1400-1409

Aronson, J., C. Floret, E. Le Floc'h, C. Ovalle, and R. Pontanier. 1993. Restoration and rehabilitation of degraded ecosystems in arid and semi-arid lands. I. A view from the south. Restoration Ecology 1(1): pp 8-17.

Yates, J.C., R. Hobbs. 2008. Woodland restoration in the Western Australian wheatbelt: a conceptual framework using a state and transition model. Restoration Ecology 5: pp 28–35.

University of Cape Town

Bakker, J.P, and Berendse, F. 1999. Constraints in the restoration of ecological diversity in grassland & heathland communities. Trends in Ecology and Evolution 4(2).

Bartolome, J.W., J.S. Fehmi, R.D. Jackson, and B. Allen-Diaz. 2004. Response of a native perennial grass stand to disturbance in California's coast range grassland. Restoration Ecology 12: pp 279–289.

Belnap, J. 1993. Recovery rates of cryptobiotic crusts: inoculant use and assessment methods. Great Basin Naturalist 53: pp 89–95.

Belnap, J. 1995. Surface disturbances: their role in accelerating desertification. Environmental Monitoring and Assessment 37: pp 39–57.

Belnap, J., and D.J. Eldridge 2003. Disturbance and recovery of biological soil crusts. Pages 364–383 in J. Belnap, and O. L. Lange, editors. Biological soil crusts: structure, function, and management. Springer- Verlag, Berlin, Germany. In Bowker, M.A. 2007. Biological Soil Crust Rehabilitation in Theory and Practice: An Underexploited Opportunity Restoration Ecology. 15(1): pp 13–23.

Belnap, J., and O. L. Lange. 2003. Biological soil crusts: structure, function, and management. Springer-Verlag, Berlin, Germany. In Bowker, M.A. 2007. Biological Soil Crust Rehabilitation in Theory and Practice: An Underexploited Opportunity Restoration Ecology 15(1): pp 13–23.

Belnap, J., and S.D. Warren. 1998. Measuring restoration success: a lesson from Patton's tank tracks. Ecological Bulletin. 79(33).

Berendse, F., M.J.M. Oomes, H.J. Altena, and W.T. Elberse. 1992. Experiments on the restoration of species-rich meadows in The Netherlands. Biological Conservation 62: pp 59–65.

Beven, K.. and P. Germann. 1982. Macropores and water flow in soils. Water Resources Research 18(5): pp 1311 – 1325.

Birch, H.F. 1958. The effect of soil drying on humus decomposition and nitrogen availability. Plant and Soil Science 10, pp 9–3.

University of Cape Town

Blackburn, W.H., R.O. Meeuwig, and C.M. Skau. 1974. A mobile infiltrometer for use on rangeland. Journal of Range Management 27: pp 322-323.

Blair N. and G.J. Crocker. 2000. Crop rotation effects on soil carbon and physical fertility of two Australian soils. Australian Journal of Soil Research 38, pp 71–84.

Blignaut, J., C. Marais, M. Rouget, M. Mander, J. Turpie, T. Klassen, and G. Preston. 2008. Making markets work for people and the environment: Employment creation from payment for eco-systems services. Online article, accessed 5th October 2010. http://www.tips.org.za/files/EGS_report_-_18_Nov.pdf

Blignaut, J., J. Aronson, M. Mander, and C. Marais. 2008. Investing in natural capital and economic development: South Africa's Drakensberg mountains. Ecological Restoration 26(2): pp 143-150.

Blignaut, J., M. Mander, R. Schulze, M. Horan, C. Dickens, C. Pringle, K. Mavundla, I. Mahlangu, A. Wilson, M. McKenzie, and S. McKean. 2010. Restoring and managing natural capital towards fostering economic development: Evidence from the Drakensberg, South Africa. Ecological Economics 69(6): pp 1313-1323.

Blumenthal, D.M., N.R. Jordan, and M.P. Russelle. 2003. Soil carbon addition controls weeds and facilitates restoration. Ecological Applications 13: pp 605–615.

Bodhinayake, W., B.C. Si, and K. Noborio. 2004. Determination of hydraulic properties in sloping landscapes from tension and double-ring infiltrometers. Vadose Zone Journal 3:3, pp 964.

Booth D.T., S.E. Cox and R.D. Berryman. 2006. Precision measurements from very-large scale aerial digital imagery. Environmental Monitoring and Assessment 112: pp 239–307.

Bowker, M.A., J. Belnap, D.W. Davidson, and H. Goldstein. 2006. Correlates of biological soil crust abundance: support for a hierarchical conceptual model. Journal of Applied Ecology 43: pp 152–163.

University of Cape Town

Bowker, M.A., J. Belnap, D.W. Davidson, and S.L. Phillips. 2005. Evidence for micronutrient limitation of biological soil crusts: importance to aridlands restoration. Ecological Applications 15: pp 1941–1951.

Bowker, M.A., S.C. Reed, J. Belnap, and S.L. Phillips. 2002. Temporal variation in community composition, pigmentation, and Fv/Fm of desert cyanobacterial soil crusts. Microbial Ecology 43: pp 13–25.

Bowker, M.A. 2007. Biological Soil Crust Rehabilitation in Theory and Practice: An Underexploited Opportunity Restoration Ecology 15(1): pp 13–23.

Boyd, J., and R.D. Simpson. 1999. Economics and biodiversity conservation options: an argument for continued experimentation and measured expectations. Science of the Total Environment 240: pp 91–105.

Briske, D.D., S.D. Fuhlendorfand, and F.E. Smeins. 2005. State-and-Transition Models, Thresholds and Rangeland Health: A Synthesis of Ecological Concepts and Perspectives. Rangeland Ecology and Management 58: pp 1-10.

Bristow K.L. 1988. The role of mulch and its architecture in modifying soil temperature. Australian Journal of Soil Research 20, pp 269–281

Brotherson, J. P., and S.R. Rushforth. 1983. Influence of cryptogamic crusts on moisture relationships of soils in Navajo National Monument, Arizona. Great Basin Naturalist 43: pp 73–78.

Bruce J.P., M. Frome, E. Haites, H. Janzen, R. Lal, and K. Paustian. 1999. Carbon sequestration in soils. Journal of Soil and Water Conservation 54: pp 382 -389.

B.P. Van Haveren. 1983. Soil Bulk Density as Influenced by Grazing Intensity and Soil Type on a Shortgrass Prairie Site. Journal of Range Management 36(5): pp. 586-588.

Butterfield, R.B., and R.F. Fisher. 1994. Untapped potential for native species reforestation. Journal of Forestry 92: pp 37–40.

University of Cape Town

Byers, J.E., K. Cuddington, C.G. Jones, T.S. Talley, A. Hastings, J.G. Lambrinos, et al. 2006. Using ecosystem engineers to restore ecological systems. Trends in ecology & evolution, 21(9): pp 493-500.

Camp, K. 1999. The Bioresource groups of KwaZulu-Natal. Cedara report no N/A99/14/ KwaZulu-Natal Department of Agriculture. Cedara, South Africa.

Carbutt, C., and T.J. Edwards. 2006. The endemic and near-endemic angiosperms of the Drakensberg alpine centre. South African Journal of Botany 72(1): pp 105-132.

Carpenter, S.R., R. De Fries, T. Dietz, H.A. Mooney, S. Polasky, W.V. Reid, and R.J. Scholes. 2006. Millennium ecosystem assessment: research needs. Science 314: pp 257-258.

Carson, R.T. 1998. Valuation of tropical rainforests: philosophical and practical issues in the use of contingent valuation. Ecological Economics 24: pp 15–29

Chapin, F. S. III. 1993. Functional role of growth forms in ecosystem and global processes. Pages 287–312. In, Eviner V.T. and Christine V. Hawkes. 2008. Embracing Variability in the Application of Plant–Soil Interactions to the Restoration of Communities and Ecosystems. Restoration Ecology 16:4, pp. 713–729

Chichilnisky, G., and G. Heal. 1998. Economic returns from the biosphere. Nature 391: pp 629–630.

Clements, F.E. 1916. Plant succession: an analysis of the development of vegetation. Carnegie Institution of Washington, Washington.

Colloff, M.J., K.R. Pullen and S.A. Cunningham. 2010. Restoration of an Ecosystem Function to Revegetation Communities: The Role of Invertebrate Macropores in Enhancing Soil Water Infiltration. Restoration Ecology 18(1): pp 65–72

Conant R.T., K. Paustian and E.T. Elliott. 2001. Grassland management and conversion into grassland: effects on soil carbon. Ecological Applications 11: pp 343-355.

University of Cape Town

Conradie, B., J. Goldin, A. Leiman, B. Standish, M. Visser. 2001. Competition policy and privatisation in the South African water industry. Cape Town, University of Cape Town, Development Policy Research Unit, 2001

Cook G.D., H.B. So and R.C. Dalal 1992. Structural degradation of two vertisols under continuous cultivation. Soil and Tillage Research 24, pp 47–64.

Costanza, R., and L. Cornwell. 1992. The 4P approach to dealing with environmental uncertainty. Environment 34: pp 12–20.

Costanza, R., R. d'Arge, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R. V. O'Neill, J. Paruelo, R. G. Raskin, P. Sutton, and M. van den Belt. 1997. The value of the world's ecosystem services and natural capital. Nature 387: pp 253–260.

Cowling, R.M., C. Hilton-Taylor, and B.J. Huntley. 1994. Patterns of plant diversity and endemism in southern Africa: an overview. Botanical diversity in southern Africa: proceedings of a Conference on the Conservation and Utilization of Southern African Botanical Diversity. National Botanical Institute, Cape Town.

Critchley, C.N.R., H.F. Adamson, B.M.L. McLean, and O.D. Davies. 2008. Vegetation dynamics and livestock performance in system-scale studies of sheep and cattle grazing on degraded upland wet heath. Agriculture, Ecosystems and Environment 128: pp 59–67.

Cross, F. B. 1993. Restoration for natural resource damages. University of Toledo Law Review 24: pp 319–334.

Daily, G. C., S. Alexander, P. R. Ehrlich, L. Goulder, J. Lubchenco, P. A. Matson, H. A. Mooney, S. Postel, S. H. Schneider, and D. Tilman. 1997. Ecosystem services: Benefits supplied to human societies by natural ecosystems. Issues in Ecology 1(2): pp 1-18.

Daily, GC. 1999. Developing a scientific basis for managing Earth's life support systems. Conservation Ecology 3(2).

Danin, A., S. Rae, M. Barbour, N. Jurjavcic, P. Conners, and E. Uhlinger. 1998. Early primary succession on dunes at Bodega Head, California. Madrono 45: pp 101–109.

University of Cape Town

Davidson, E.A., C.J.R. de Carvalho, I.C.G. Vieira, R.D. Figueiredo, P. Moutinho, F.Y. Ishida, dos M.T.P Santos, J.B. Guerrero, K. Kalif, and R.T. Saba 2004. Nitrogen and phosphorus limitation of biomass growth in a tropical secondary forest. Ecological Applications 14: pp 150–163.

Didham, R.K., C.H. Watts, D.A. Norton. 2005. Are systems with strong underlying abiotic regimes more likely to exhibit alternative stable states? Oikos, 110(2), pp 409–416

Dirksen, C. 2000. Unsaturated Hydraulic Conductivity. Pages 183–239 in Smith K.A. & Mullins C.E., editors. Soil and Environmental Analysis Physical Methods, second edition. Marcel Dekker inc., New York – Basel.

Du Plessis H.M. and I. Shainberg. 1985. Effect of exchangeable sodium and phosphogypsum on the hydraulic properties of several South African soils. South African Journal of Plant and Soil 2: pp 179–186.

Du Preez C.C. and H.A. Snyman. 1993. Organic matter content of a soil in a semi-arid climate with three long-standing veld conditions. African Journal of Range and Forage Science 10: pp 108–110.

Du Toit M.E. and C.C. Du Preez .1995. Effect of cultivation on the nitrogen fertility of selected dryland soils in SouthAfrica. Soil Science Society of America journal 12: pp 73–81.

Dunne, T., D. Western, & W.E. Dietrich. 2011. Effects of cattle trampling on vegetation, infiltration, and erosion in a tropical rangeland. Journal of Arid Environments 75(1): pp 58-69.

Dysterhuis, E.J. 1949. Condition and management of rangeland based on quantitative ecology. Journal of Rangeland Management 2: pp 104-105.

Edmond, D.B. 1962. Effects of treading pasture in summer under different soil moisture levels. New Zealand Journal of Agricultural Research 5: pp 389-395.

University of Cape Town

Edwards J.H., C.W. Wood, D.L. Thurlow and M.E. Ruf. 1992. Tillage and crop rotation effects on fertility status of a hapludult soil. Soil Science Society of America journal 56, pp 1577–1582.

Ehrenfeld, J. G., B. Ravit, and K. Elgersma. 2005. Feedback in the plantsoil system. Annual Review of Environment and Resources 30: 75–115.

Eldridge, D.J., E. Zaady, and M. Shachack. 2002. Microphytic crusts, shrub patches, and water harvesting in the Negev desert: the Shikim system. Landscape Ecology 17: pp 587–597.

Elliott JA, A.A. Efetha. 1999. Influence of tillage and cropping system on soil organic matter, structure and infiltration in a rolling landscape. Canadian Journal of Soil Science:79(3): pp 457-463.

Ellison, L. 1960. Influence of grazing on plant succession of rangelands. The Botanical Review 26: pp1-78.

Enserink, M. 1999. Plan to quench the Everglade's thirst. Science 285: pp 180.

Evans, R. D., and J.R. Ehleringer. 1993. A break in the nitrogen cycle of arid lands? Evidence from d15 N of soils. Oecologia 94: pp 314–317.

Everson C.S., and N.M. Tainton. 1984. The effects of thirty years of burning in the highland sourveld of Natal. Journal of the Grassland Society of Southern Africa. 1(3): pp 15-20.

Everson, C.S., and G.P.Y. Clarke. 1987. A comparison of six methods of botanical analysis in the montane grasslands of natal. Vegetation 73(1): pp 47-51.

Everson, T.M., C.S. Everson, and K.D. Zuma. 2007. Community Based Research on the influence of rehabilitation techniques on the management of degraded catchments. Online article, accessed October 5th 2010. http://www.wrc.org.za/Knowledge%20Hub%20Documents/Research%20Reports/1316%20web.pdf

Everson, T.M., G.P.Y. Clarke, and C.S. Everson. 1990. Precision in monitoring plant species composition in montane grasslands. Plant Ecology 88(2): pp 135-141.

University of Cape Town

Everson, T.M., R.I. Yeaton, and C.S. Everson. 2009. Seed dynamics of Themeda triandra in the montane grasslands of South Africa. African Journal of Range and Forage Science 26:1.

Eviner V.T. and C.V. Hawkes. 2008. Embracing Variability in the Application of Plant–Soil Interactions to the Restoration of Communities and Ecosystems. Restoration Ecology (16)4: pp 713–729.

Eviner, V. T., and F. S. Chapin III. 2001. Plant species provide vital ecosystem functions for sustainable agriculture, rangeland management, and restoration. California Agriculture 55: pp 54–59.

Fearnehough, W., M.A. Fullen, D.J. Mitchell, I.C. Trueman, and J. Zhang. 1998. Aeolian deposition and its effect on soil and vegetation changes on stabilized desert dunes in northern China. Geomorphology 23: pp 171–182.

Feldpausch, T. R., M.A. Rondon, E.C.M. Fernandes, S.J. Riha, and E. Wandelli. 2004. Carbon and nutrient accumulation in secondary forests regenerating on pastures in central Amazonia. Ecological Applications 14: pp164–176.

Francis G.S., F.J. Tabley and K.M. White. 2001. Soil degradation under cropping and its influence on wheat yield on a weakly structured New Zealand silt loam. Australian Journal of Soil Research 39: pp 291–305.

Free, G.R., G.M. Browning, and G.W. Musgrave. 1940. Relative infiltration and related physical characteristics of certain soils. USDA Technical Bulletin 729. US Department of Agriculture.

Frenkel H., M.V. Fey, G.H. Goodall and W.B. Russell .1989. Effect of soil surface amendments on runoff and erosion from simulated rain applied to sesquioxidic soil. South African Journal of Plant and Soil. 6, pp 197–202.

Gamougoun, N.D., R.P. Smith, M.K. Wood, and R.D. Pieper. 1984. Soil, vegetation, and hydrologic responses to grazing management at Fort Stanton, New Mexico. Journal of Range Management 37: pp 538-541.

University of Cape Town

Geist, C., and S. M. Galatowitsch. 1999. Reciprocal model for meeting ecological and human needs in restoration projects. Conservation Biology 13: pp 970–979.

Gifford, G.F., and R.H. Hawkins. 1976. Grazing systems and watershed management: A look at the record. Journal of Soil and Water Conservation 31: pp 281-283.

Gifford, G.F., and R.H. Hawkins. 1978. Hydrologic impact of grazing on infiltration: A critical review. Water Resources Research. 14: pp 305-313.

Global Environmental Facility (GEF). 1998. Project implementation review of the global environmental facility 1997. Online Article, accessed 21 October 2010, http:/www.gefweb.org/assets/images/report.doc.

Goodloe, S. 1969. Short-duration grazing in Rhodesia. Journal of Range Management. 22: pp 369-373.

Gordon, I.J., P. Duncan, P. Grillas, and T. Lecomte. 1990. The use of domestic herbivores in the conservation of the biological richness of European wetlands. Bulletin d'Ecologie 21: pp 49–60.

Gregorich E.G., C.F. Drury and J.A. Baldock. 2001. Changes in soil carbon under long-term maize in monoculture and legume-based rotation. Canadian Journal of Soil Science 81: pp 21-31.

Gretarsdottir, J., A.L. Aradottir, V. Vandvik, E. Heegaard, and H.J.B. Birks. 2004. Long term effects of reclamation treatments on plant succession in Iceland. Restoration Ecology 12: pp 268–278.

Grime, J.P. 1977. Evidence for the Existence of Three Primary Strategies in Plants and Its Relevance to Ecological and Evolutionary Theory. The American Naturalist 111:982 pp 1169-1194

Guinon, M. 1989. No free lunch. Restoration and Management Notes . In Holl K.D. and R.B. Howarth. 2000. Paying for Restoration. Restoration Ecology Vol. 8 No. 3, pp. 260–267

Guo L.B. and R.M. Gifford. 2002. Soil carbon stocks and land use change: a meta-analysis. Global Change Biology 8: pp 345-360.

University of Cape Town

Håkansson, I. and J.Lipiec. 2000. A review of the usefulness of relative bulk density values in studies of soil structure and compaction. Journal of Soil and Tillage Research, 53(2): pp71-85.

Hall, C.A.S. 1992. Economic development or developing economics: what are our priorities? pp 101–126 in in Holl K.D. and R.B. Howarth. 2000. Paying for Restoration. Restoration Ecology Vol. 8 No. 3, pp. 260–267.

Hancock, D.W., R.C Lacy, L. Stewart, R.S. Tubbs, J.M. Kichler, T.W. Green, R Hicks,. 2010. The management and use of bahiagrass. The University of Georgia Cooperative Extension Bulletin, 1362.

Hardy, M.B., and N.M. Tainton. 2007. Towards a technique for determining basal cover in tufted grasslands. African Journal of Range and Forage Science 10(2): pp 77.

Hartnett, D.C., K.R. Hickman, & L.E.F. Walter. 1996. Effects of bison grazing, fire, and topography on floristic diversity in tallgrass prairie. Journal of Range Management, 49, pp 413–420.

Hassink J. 1997. The capacity of soils to preserve organic C and N by their association with clay and silt particles. Plant and Soil 191: pp 77- 87.

Heneghan, L., S.P. Miller, S. Baer, M.A. Callaham, J. Montgomery, M. Pavao-Zuckerman, C.M. Rhoades, & Richardson, S. 2008. Integrating Soil Ecological Knowledge into Restoration Management. Restoration Ecology 16(4): pp 608–617.

Heyde, J. M. 1995. Is contingent valuation worth the trouble? Chicago Law Review 62: 331–362.

Hibbert, A.R. 1983. Water Yield Improvement Potential by Vegetation Management On Western Rangelands. Journal of the American Water Resources Association 19: pp 375–38.

Hobbs, R. T., and J.A. Harris. 2001. Restoration ecology: repairing the Earth's ecosystems in the new millennium. Restoration Ecology 9: pp 239–246.

University of Cape Town

Hodacova, D., and K. Prach. 2003. Spoil heaps from brown coal mining: technical reclamation vs. spontaneous re-vegetation. Restoration Ecology 11: pp 385–391.

Holechek, J.L., T.T. Baker, J.C. Boren, and D. Galt. 2006. Grazing impacts on rangeland vegetation: what we have learned. Rangelands 28: pp 7–13.

Holl K.D. and R.B. Howarth. 2000. Paying for Restoration. Restoration Ecology 8:(3) pp 260–267.

Holloran, P. 1996. Greening the Golden Gate. Restoration and Management Notes 14: pp 112–123.

Hongo, A., S. Matsumoto, H. Takahashi, H.Y. Zou, J.M. Cheng, H.Y. Jia, and Z.Y. Zhao. 1995. Effect of exclosure and topography on rehabilitation of overgrazed shrub-steppe in the Loess Plateau of Northwest China. Restoration Ecology 3: pp 18–25.

Hoveland, C.S. 2000. Achievements in Management and Utilization of Southern Grasslands. Journal of Range Management, 53(1): pp. 17-22

Hungate B.A., E.A. Holland, R.B. Jackson, F.S. Chapin, H.A. Mooney, and C.B. Field. 1997. The fate of carbon in grasslands under carbon dioxide enrichment. Nature 388: pp 576-579.

Hunt R.E.J., J.H. Everitt, J.C. Ritchie, M.S. Moran, D.T. Booth, G.L. Anderson, Clark P.E. and M.S. Seyfried. 2003 .Applications and research using Moran remote sensing for rangeland management. Photogrammetric Engineering and Remote Sensing 69(6): pp 675–693.

Hurt C.R. 1989. The Relationship between range condition score and long-term stocking rates applied to two vegetation types in Natal. M.Sc. Agric. thesis. University of Natal, Pietermaritzburg.

Huston, M. A. 2004. Management strategies for plant invasions: manipulating productivity, disturbance, and competition. Diversity and Distribution, 10: pp 167–178.

IPCC. 2001. Climate change 2001: the scientific basis, contribution of working group 1 to the third assessment report of the intergovernmental panel on climate change. Houghton

University of Cape Town

J.T., Ding Y, Griggs D.J., Noguer M, van der Linden P.J., Dai X, Maskell K, Johnson C.A., eds. Cambridge, UK: Cambridge University Press.

Jackson R.B., J. Canadell, J.R. Ehleringer, H.A. Mooney, O.E. Sala and E.D. Schulze. 1996. A global analysis of root distributions for terrestrial biomes. Oecologia 108: pp 389-411.

Jenkinson D.S. 1990. The turnover of organic carbon and nitrogen in soil. Philosophical Transactions of the Royal Society of London Series B - Biological Sciences 329: pp 361-368.

Jenkinson D.S. and J.H. Rayner 1977. The turnover of soil organic matter in some of the Rothamsted classical experiments. Soil Science. 123: pp 298–305.

Joel, A. 2000. Application of two methods to determine hydraulic conductivity with disc permeameters on sloping land. European Journal of Soil Science 51: pp 93–98.

Jones, M.B. and A. Donnelly. 2004. Carbon Sequestration in Temperate Grassland Ecosystems and the Influence of Management, Climate and Elevated CO^2. New Phytologist (164)3: pp 423-439.

Jordan, W., M.E. Giplin, and J.D. Aber, editors. 1987. Restoration ecology: a synthetic approach to ecological research. Cambridge University Press, Cambridge, United Kingdom.

Kalmbacher, R.S., F.G. Martin, 1998. Effect of flooding on seed germination and emergence of three pasture grasses. Soil & Crop Science Society of Florida Proceedings, 57: pp73–77.

Keane, R.M and M.J. Crawley. 2002. Exotic plant invasions and the enemy release hypothesis. TRENDS in Ecology & Evolution, 17:.

Killick, D.J.B. 1963. Ecology of the Cathedral Peak Area, Memoirs of the Botanical Survey of South Africa, Number 34, Botanical Research Institute, Pretoria.

University of Cape Town

King, E.G., and R.J. Hobbs. 2006. Identifying linkages among conceptual models of ecosystem degradation and restoration: towards an integrative framework. Restoration Ecology 14: pp 369–378.

Knetsch, J. L. 1990. Environmental policy implications of disparities between willingness to pay and compensation demanded measures of values. Journal of Environmental Economics and Management 18: 227–237.

Knoll, G., and H.H. Hopkins. 1959. The effects of grazing and trampling upon certain soil properties. Transactions of the Kansas Academy of Science 62: pp 221-231.

Koo B., Barton C., and D.Adriano 2006. Evaluation of Bahiagrass (*Paspalum notatum*) as a Vegetative Cover for a Landfill Containing Coal Combustion Waste. pp 225 - 231 in Coal combustion byproducts and environmental issues. Sajwan K.S., Twardowska I. & Punshon T. (Eds). Springer Verlag, USA.

Kova, R.P. 2004. Natural recovery of human-made deposits in landscape (biotic interactions and ore/ash-slag artificial ecosystems). Restoration Ecology (14)4: pp 676.

Krueger-Mangold, J.M., R.L. Sheley, and T.J. Svejcar. 2006. Toward ecologically-based invasive plant management on rangeland. Weed Science 54: pp 597–605.

Kurina, L.M., and P.M. Vitousek. 1999. Controls over the accumulation and decline of a nitrogen-fixing lichen, Stereocaulon vulcani, on young Hawaiin lava flows. Journal of Ecology 87: pp 784–799.

Lange, O.L., A. Meyer, H. Zellner, and U. Heber. 1994. Photosynthesis and water relationships of lichen soil crusts: field measurements in the coastal fog zone of the Namib desert. Functional Ecology 8: pp 253–264.

Levine, J.M., E. Pachepsky, B.E. Kendall, S.G. Yelenik, and J.H.R. Lambers. 2006. Plant-soil feedbacks and invasive spread. Ecology letters 9: pp 1005–1014.

Levy, E.B. & E.A. Madden. 1933. The point method of pasture analysis. New Zealand Journal of Agriculture. 46: pp 267-279.

University of Cape Town

Lilliefors, H. 1967. On the Kolmogorov–Smirnov test for normality with mean and variance unknown. Journal of the American Statistical Association, Vol. 62: pp 399–402.

Lonsdale, W.M. 1999. Global patterns of plant invasions and the concept of invasibility. Ecology 80:pp 1522–1536.

Lord, E. 1947. The Use of Range in Place of Standard Deviation in the t-Test Biometrika Vol. 34, 1: pp 41-67.

Lull, H.W. 1959. Soil compaction on forest and range lands. USDA Misc. Pub. 768. In Warren, S.D., T.L. Thurow, W.H. Blackburn, and N.E. Garza. 1986.The Influence of Livestock Trampling under Intensive Rotation Grazing on Soil Hydrologic Characteristics. Journal Of Range Management 39:6.

Maestre, F.T., A. Escudero, I. Martinez, C. Guerrero, and A. Rubio. 2005. Does spatial patterning matter to ecosystem functioning? Insights from biological soil crusts. Functional Ecology 19: pp 566–573.

Maestre, F. T., and J. Cortina. 2002. Spatial patterns of surface soil properties and vegetation in a Mediterranean semi-arid steppe. Plant and Soil 241: pp 279–291.

Maestre, F. T., S. Bautista, J. Cortina, and J. Bellot. 2001. Potential for using facilitation by grasses to establish shrubs on a semiarid degraded steppe. Ecological Applications 11: pp 1641–1655.

Mando, A., L. Brussard, and L. Stroosnijder. 1994. Termite and mulch mediated rehabilitation of vegetation on crusted soil in West Africa. Restoration Ecology 7: pp 33–41.

Marais, C., and A.M. Wannenburgh. 2008. Restoration of water resources (natural capital) through the clearing of invasive alien plants from riparian areas in South Africa - costs and water benefits. South African Journal of Botany 74(3): pp 526-537.

Marshall, W.A., M.O. Chalmers. 1997. Airborne Dispersal of Antarctic Terrestrial Algae and Cyanobacteria. Ecography 20(6): pp 585-594.

University of Cape Town

Mazor, G., G.J. Kidron, A. Vonshak, and A. Abeliovich. 1996. The role of cyanobacterial exopolysaccharides in structuring desert microbial crusts. FEMS Microbiology Ecology 21:pp 121–130.

McCalla, G.R., W.H. Blackburn, and L.B. Merrill. 1984a. Effects of live- stock grazing on infiltration rates, Edwards Plateau of Texas. Journal of Range Management. 37: pp 265-269.

McCalla, G.R., W.H. Blackbum, and L.B. Merrill. 1984b. Effects of livestock grazing on sediment production, Edwards Plateau of Texas. Journal of Range Management. 37: pp 291-294.

McIver, J. and L. Starr. 2001. Restoration of degraded lands in the interior Columbia River basin: passive vs. active approaches. Forest Ecology and Management 153(1-3): pp 15-28.

Miller, M. E. 2005. The structure and functioning of dryland ecosystems - conceptual models to inform long-term ecological monitoring USGS SIR 2005-5197. United States Geological Survey, Reston, Virginia. In Bowker, M.A. 2007. Biological Soil Crust Rehabilitation in Theory and Practice: An Underexploited Opportunity Restoration Ecology 15(1): pp 13–23.

Mills A.J. and M.V. Fey. 2003. Declining soil quality in South Africa: effects of land use on soil organic matter and surface crusting. South African Journal of Science 99: pp 429-436.

Mortimer, S.R., J.A.Hollier, and V.K.Brown, 1998. Interactions between plant and insect diversity in the restoration of lowland calcareous grasslands in southern Britain. Journal of Applied Vegetation Science, 1(1): pp101-114

Mucina, L., and M.C. Rutherford. 2006. The vegetation of south Africa, Lesotho and Swaziland. South African National Biodiversity Institute (SANBI). Cape Town, South Africa.

University of Cape Town

Muller, S., T. Dutoit, D. Alard, and F. Grevilliot. 1998. Restoration and Rehabilitation of Species-Rich Grassland Ecosystems in France: a Review. Restoration Ecology 6(1): pp 94-101.

Mworia J.K., W.N Mnene., D.K. Musembi and R.S. Reid. 1997. Resilience of soils and vegetation subjected to different grazing intensities in a semi-arid rangeland of Kenya. African Journal of Range and Forage Science. 14, pp 26–31.

Ninot, J.M., P. Herrero, A. Ferre´, and R. Guardia. 2001. Effects of reclamation measures on plant colonization on lignite waste in the eastern Pyrenees, Spain. Applied Vegetation Science 4: pp 29–34.

Numata, M. 1992. The degradation of grassland ecosystems and their recovery. In Muller, S., T. Dutoit, D. Alard, F. Grévilliot. 1998. Restoration and Rehabilitation of Species-Rich Grassland Ecosystems in France: a Review. Restoration Ecology 6:1, pp. 94–101.

Oliver, M.J., B.D. Mishler, and J.E. Quisenberry. 1993. Comparative measures of desiccation-tolerance in the Tortula ruralis complex. I. Variation in damage control and repair. American Journal of Botany 80: pp 127–136.

Palmer M.A., R.F. Ambrose, N.L.R. Poff. 1997. Ecological theory and community restoration ecology. Restoration Ecology 5(4): pp 291–300.

Papanastasis, V. P., M.D. Yiakoulaki,. M. Decandia, and O. Dini-Papanastasi. 2008. Integrating woody species into livestock feeding in the Mediterranean areas of Europe. Animal Feed Science and Technology pp 140: pp 1–17.

Papanastasis, V.P. 2009. Restoration of Degraded Grazing Lands through Grazing Management: Can It Work? Restoration Ecology 17:4, pp 441–445.

Paul E.A., R.F. Follett, S.W. Leavitt, A. Halvorson, G.A. Peterson and D.J. Lyon. 1997a. Radiocarbon dating for determination of soil organic matter pool sizes and dynamics. Soil Science Society of America Journal 61: pp 1058-1067.

Paul EA, K. Paustian, H.P. Collins, U. Schulthess, and G.P. Robertson. 1997b. Soil organic matter in temperate agroecosystems. Boca Raton, FL, USA: CRC Press. In Jones, M.B.

University of Cape Town

and A. Donnelly. 2004. Carbon Sequestration in Temperate Grassland Ecosystems and the Influence of Management, Climate and Elevated CO2. New Phytologist (164)3: pp 423-439.

Perroux K.M. and I. White. 1987. Designs for Disc Permeameters. Soil Science Society of America 52(5): pp 1205-1215.

Perrow,M.R. and A.J. Davy, editors. 2002. Handbook of ecological restoration. Cambridge University Press, Cambridge, United Kingdom.

Post W.M., K.C. Kwon. 2000. Soil carbon sequestration and land-use change: processes and potential. Global Change Biology 6: pp 317-327.

Poulton, E.C. 1975. Observer bias. Applied Ergonomics, 6(1): pp 3-8.

Prach, K. 1993. Vegetational changes in a wet meadow complex, South Bohemia, Czech Republic. Folia Geobotanica Phytotaxonomica Praha 28: pp 1–13. In Muller, S., T. Dutoit, D. Alard, and F. Grevilliot. 1998. Restoration and Rehabilitation of Species-Rich Grassland Ecosystems in France: a Review. Restoration Ecology 6(1): pp 94-101.

Prach, K., and R.J. Hobbs. 2008. Spontaneous Succession versus Technical Reclamation in the Restoration of Disturbed Sites. Restoration Ecology 16(3): pp 363-366.

Prentice I.C., G.D. Farquhar, M.J.R. Fasham, M.L. Goulden, M. Heimann, V.J. Jaramillo, H.S. Kheshgi, C. Le Quere, R.J. Scholes, D.W.R. Wallace. 2001. The carbon cycle and atmospheric carbon dioxide. In: Jones, M.B. and A. Donnelly. 2004. Carbon Sequestration in Temperate Grassland Ecosystems and the Influence of Management, Climate and Elevated CO2. New Phytologist, 164:3, pp. 423-439.

Pywell, R.F., J.M. Bullock, D.B. Roy, L.I.Z. Warman, K.J. Walker, and P. Rothery. 2003. Plant traits as predictors of performance in ecological restoration. Journal of Applied Ecology 40: pp 65–77.

Rauzi, F. and C.L. Hanson. 1966. Water intake and runoff as affected by intensity of grazing. Journal of Range Management. 19: pp 351-356.

University of Cape Town

Rayburn, W.R., R.N. Mack, and B. Metting. 1982. Conspicuous algal colonization of the ash from Mount St. Helens. Journal of Phycology 18: pp 537–543.

Reed, M.J., and R.A. Peterson. 1961. Vegetation, soil, and cattle responses to grazing on Northern Great Plains range. USDA Technical Bulletin 1252.

Reeder J.D. and G.E. Schuman. 2002. Influence of livestock grazing on C sequestration in semi-arid mixed-grass and short-grass rangelands. Environmental Pollution 116: pp 457-463.

Reeder, J.D., G.E. Schuman, J.A. Morgan, and D.R. Lecain. 2004. Response of organic and inorganic carbon and nitrogen to long-term grazing of the shortgrass steppe. Environmental Management, (33)4: pp 485-495.

Reganold, J.P., Palmer, A.S., 1995. Significance of gravimetric versus volumetric measurements of soil quality under biodynamic, conventional, and continuous grass management. Journal of Soil & Water Conservation, 50(3): pp 298-305.

Rehounkova, K., and K. Prach. 2007. Spontaneous vegetation succession in gravel-sand pits: a potential for restoration. Restoration Ecology, 16:2 pp 305.

Reicosky D.C., W.A. Dugas and H.A. Torbert .1997. Tillage-induced soil carbon dioxide loss from different cropping systems. Soil and Tillage Research. 41, pp 105–118.

Reicosky, D.C., W.D. Kemper, G.W. Langdale, C.L. Douglas, P.E. Rasmussen. 1995. Soil organic matter changes resulting from tillage and biomass production. Journal of Soil and Water Conservation. 50 (3).

Rein, F. A. 1999. An economic analysis of vegetative buffer strip implementation. Coastal Zone Management 27: pp 377–390.

Reynolds, C.M., D.C. Wolf, T.J. Gentry, L.B. Perry, C.S. Pidgeon, B.A. Koenen, H.B. Rogers, C.A. Beyrouty, 1999. Plant enhancement of indigenous soil microorganisms: a low-cost treatment of contaminated soils. Polar Record 35(192): pp 33–40.

University of Cape Town

Rhoades, E.D., L.F. Locke, H.M. Taylor, and E.H. Mcllvain. 1964. Water intake on a sandy range as affected by 20 years of differential cattle stocking rates. Journal of Range Management. 17: pp 185-190.

Ritchie, J.T. 1998. Soil water balance and plant water stress. Systems approaches for sustainable agricultural development, 7: pp 41-54.

Robles M.D. and I.C. Burke .1998. Soil organic matter recovery on conservation reserve program fields in southeastern Wyoming. Soil Science Society of America Journal. 62: pp 725–730.

Ruiz-Jaen, M.C., & T Mitchell-Aide. 2005. Restoration Success: How Is It Being Measured? Restoration Ecology, 13(3): pp 569-577.

Savory A. 1983. The Savory grazing method or holistic resource management. Rangelands 5: pp 155-159.

Savory, A., and S.D. Parsons. 1980. The Savory grazing method. Rangelands 2: pp 234-237.

Scarlett, N. 1994. Soil crusts, germination and weeds—issues to consider. Victorian Naturalist 111: pp 125–130.

Schlesinger W.H., A.D. Abrahams, A.J. Parsons, J. Wainwright. 1999. Nutrient losses in runoff from grassland and shrubland habitats in southern New Mexico: I. Rainfall simulation experiments. Biogeochemistry 45: pp 21–34.

Schlesinger W.H., T.J. Ward, J. Anderson. 2000. Nutrient losses in runoff from grassland and shrubland habitats in southern New Mexico: II. Field plots. Biogeochemistry 49: pp 69–86

Schoenholtz S.H., H. Van Miegroet, J.A. Burge. 2000. A review of chemical and physical properties as indicators of forest soil quality: challenges and opportunities. Forest Ecology and Management 138: pp 335-356.

Schulze, R.J. 1997. South African Atlas of Agrohydrology and Climatology. Report TT82/96. Water Research Commission, Pretoria.

University of Cape Town

Seastedt T.R., C.C. Coxwell, D.S. Ojima and W.J. Parton. 1994. Controls of plant soil carbon in a semi-humid temperate grassland. Ecological Applications. 4: pp 344–353.

Seitlheko E.M., B.L. Allen and D.B. Wester. 1993. Effect of three grazing intensities on selected soil properties in semi-arid west Texas. African Journal of Range and Forage Science. 10: pp 82–85.

SER (Society for Ecological Restoration International Science & Policy Working Group). 2004. The SER International Primer on Ecological Restoration Online Article, accessed October 2010. http//www.ser.org.

Shainberg I. and G.J. Levy. 1994. Organic polymers and soil sealing in cultivated soils. Soil Science 158, pp 267–273.

Sluis, W.J. 2002. Patterns of Species Richness and Composition in Re-Created Grassland. Restoration Ecology, 10(4): pp 677 – 684.

Smoliak S., J.F. Dormaar., and A. Johnston. 1972. Long-term grazing effects on Stipa-boiuteloua prairie soils. Journal of Rangeland Management. 25: pp 246-250.

Sombroek W.G. 1986. Introduction to the symposium. In Assessment of Soil Surface Sealing and Crusting, eds F. Callebaut, D. Gabriels and M. De Boodt, pp. 1–7. State University of Ghent, Ghent. In Mills A.J. and M.V. Fey. 2003. Declining soil quality in South Africa: effects of land use on soil organic matter and surface crusting. South African Journal of Science 99: 429-436.

Soussana J.F., P. Loiseau, N. Vuichard, E. Ceschia, J. Balesdent, T. Chevallier and D. Arrouays. 2004. Carbon cycling and sequestration opportunities in temperate grasslands. Soil Use and Management 20: pp 219-230.

Stern R., M. Ben-Hur and I. Shainberg. 1991. Clay mineralogy effect on rain infiltration, seal formation and soil losses. Soil Science. 152, pp 455–462.

Suding K.N., K.L. Gross, G.R. Houseman. 2004. Alternative states and positive feedbacks in restoration ecology. Trends in Ecology & Evolution, 19(1): pp 46–53.

Sycholt, A. 2003. A Guide to the Drakensberg. Struik, Cape Town.

University of Cape Town

Tainton, N. 1999. Veld Management in South Africa. University of Natal Press, Pietermaritzberg.

Theron J.J. 1951. The influence of plants on the mineralisation of nitrogen and the maintenance of organic matter in the soil. The Journal of Agricultural Science. 41, pp 289–296.

Thornes, J.B. 2007. Modelling soil erosion by grazing: Recent developments and new approaches. Geographical Research 45: pp 13–16.

Thurow, T.L., W.H. Blackburn and C.A. Taylor Jr. 1986. Hydrologic Characteristics of Vegetation Types as Affected by Livestock Grazing Systems, Edwards Plateau, Texas. Journal of Range Management, 39(6): pp 505-509.

Tisdall J.M. and J.M. Oades. 1982. Organic matter and water-stable aggregates in soils. European Journal of Soil Science. 33, pp 141–163.

Tongway, D. 2003. Landscape Function Analysis: a System for Monitoring Landscape Function. Pp 7. Methodologies and Indicators for the Evaluation of Restoration Projects. Proceedings of the first REACTION workshop. Alicante, Spain.

Tongway, D. J., and N. L. Hindley. 2004. Landscape function analysis: procedures for monitoring and assessing landscapes; With special reference to Minesites and Rangelands. CSIRO Sustainable ecosystems, Australia.

Toth, L.A., D.A. Arrington, M.A. Brady, and D.A. Muszick. 1995. Conceptual evaluation of factors potentially affecting restoration of habitat structure within the channelized Kissimmee River ecosystem. Restoration Ecology 3:pp 160–180.

Tringham, T.K., W.C. Krueger, and P.L. Shaver. 2003. State amd transition modelling: an ecological process approach. Journal of Range Management, 56: pp 106-113.

Trollope W.S.W., Trollope L.A., and O.H.J. Bosch 1990. Veld and pasture management terminology in Southern Africa. Journal of the Grassland Society of Southern Africa, 71: pp 52-61.

University of Cape Town

Turpie, J. K., C. Marais, and J. N. Blignaut. 2008. The working for water programme: Evolution of a payments for ecosystem services mechanism that addresses both poverty and ecosystem service delivery in south africa. Ecological Economics 65(4): pp 788-798.

Uchida, T., N. Ohte, A. Kimoto, T. Mizuyama, and L. Chnaghua. 2000. Sediment yield on a devastated hill in southern China: effects of microbiotic crust on surface erosion process. Geomorphology 32: pp 129–145.

Uys, R.G., W.J. Bond, & T.M. Everson, 2004. The effect of different fire regimes on plant diversity in southern African grasslands. Biological Conservation, 118: pp 489-499.

Van Andel, J., J. Aronson. 2006. Restoration ecology: the new frontier. Wiley-Blackwell publishers, Oxford, UK.

Van Poollen, H.W., and J.R. Lacey. 1979. Herbage response to grazing systems and stocking intensities. Journal of Range Management. 32: pp 250-253.

Van Wilgen, B.W., R.M. Cowling, and C.J. Burgers. 1996. Valuing of ecosystem services. Bioscience 46: pp 184–189.

Vinton, M.A., and E.M. Goergen. 2006. Plant-soil feedbacks contribute to the persistence of Bromus inermis in tallgrass prairie. Ecosystems 9: pp 967–976.

Walker, L.R., and R. del Moral. 2003. Primary succession and ecosystem rehabilitation. Cambridge University Press, Cambridge, United Kingdom. In Prach, K and Hobbs, JR. 2008. Succession versus Technical Reclamation in the Restoration of Disturbed Sites Restoration Ecology 16:3, 363–366.

Walkley, A. Black, A.I. 1934. An Examination of the Degtjareff Method for Determining Soil Organic Matter, and A Proposed Modification of the Chromic Acid Titration Method. Soil Science: 37: pp 29-38.

WallisDeVries, M.F., P. Poschlod, J.H. Willems. 2004. Challenges for the conservation of calcareous grasslands in North Western Europe: integrating the requirements of flora and fauna. Biological Conservation, 104(3): pp 265 – 273.

University of Cape Town

Warren, S.D. T.L. Thurow, W.H. Blackburn, and N.E. Garza. 1986.The Influence of Livestock Trampling under Intensive Rotation Grazing on Soil Hydrologic Characteristics. Journal Of Range Management 39:6.

Wassenaar, T.D., S.M. Ferreira, and R.J. Van Aarde. 2007. Flagging aberrant sites and assemblages in restoration projects. Restoration Ecology 15: pp 68–76.

Watson K.W. and R.J. Luxmoore. 1986. Estimating Macroporosity in a Forest Watershed by use of a Tension Infiltrometer. Soil Science Society of America Soil Science Society of America 50(3): pp 578-582.

Westoby M., B. Walker, and I. Noy-Meir. 1989. Opportunistic Management for Rangelands Not at Equilibrium. Journal of Range Management, Vol. 42, No. 4, pp. 266-274

Whisenant, S.G. 1999. Repairing damaged wildlands: a process-oriented, landscape-scale approach. Cambridge University Press, Cambridge, United Kingdom.

Whisenant, S.G., T.L. Thurow, and S.J. Maranz. 1995. Initiating autogenic restoration on shallow semiarid sites. Restoration Ecology 3: pp 61–67.

Wilkinson, M.T., P.J. Richards, and G.S. Humphreys. 2009. Breaking ground: pedological, geological, and ecological implications of soil bioturbation. Earth-Science Reviews, 97(1-4): pp 257-272.

Wilson, M. A., and S.R. Carpenter. 1999. Economic valuation of freshwater ecosystem services of the United States: 1971-1997. Ecological Applications 9: pp 772–783.

Wood, M.K., and W.H. Blackburn. 1981a. Grazing systems: Their influence on infiltration rates in the Rolling Plains of Texas. Journal of Range Management. 34: pp 331-335.

Wood, M.K., and W.H. Blackburn. 1981b. Sediment production as influenced by livestock grazing in the Texas Rolling Plains. Journal of Range Management. 34: pp 228-231.

Wu, L., L. Pan, M. Robertson, and P. Shouse. 1997. Numerical evaluation of ring infiltrometers under various soil conditions. Soil Science. 162: pp 771-777.

University of Cape Town

Wunder, S. 2005. Payments for environmental services: some nuts and bolts. Center for International Forestry Research, Bogor. Online Article. Accessed October 2010. http://www.cifor.cgiar.org/publications/pdf_files/OccPapers/OP-42.pdf

Xia H.P. 2004. Ecological rehabilitation and phytoremediation with four grasses in oil shale mined land. Chemosphere, 54: pp 345–353.

Xia, H.P., Shu, W.S., 2001. Resistance to and uptake of heavy metals by *Vetiveria zizanioides* and *Paspalum notatum* from lead/zinc mine tailings. Acta Ecol. Sinica 21, pp 1121–1129.

Yates, C.J., and R.J. Hobbs. 1997. Woodland restoration in the Western Australian wheatbelt: a conceptual framework using a state and transition model. Restoration Ecology 5: pp 28–35.

Young, T. P. 2000. Restoration ecology and conservation biology. Biological Conservation 92: pp 73–83.

Youngs G.S. 2000. Hydraulic Conductivity of Saturated soils. Pages 141–183 in Smith K.A. & Mullins C.E., editors. Soil and Environmental Analysis Physical Methods, second edition. Marcel Dekker inc., New York, Basel.

University of Cape Town