ASSESSING SOIL CARBON AND CARBON DIOXIDE EFFLUXES UNDER DIFFERENT VEGETATION COVER CONDITIONS IN THE EASTERN CAPE PROVINCE, SOUTH AFRICA

By

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Abstract

Albany thicket is prevalent in the Eastern Cape Province of South Africa. Its spread has diminished through overgrazing and heavy browsing by animals, land clearance and urban expansion. The result is highly degraded land characterized by invasion of alien species. There is a wealth of documented evidence on the high carbon sequestration ability of thicket biome, but not much has been done to assess its effect on carbon dioxide emissions from the soil. Given that the concentration of atmospheric greenhouse gases has been constantly rising since the industrial era, it is imperative to assess the influence of thicket biome as a source or sink of these gases. There is evidence of shifts in the climate in southern Africa as reflected by changes in rainfall patterns, increased temperatures, recurrent droughts and fires. As such, the historical rainfall variability in an Albany thicket region and its interaction with the temporal land use / cover changes was studied. This served to give some background information about the study area for more detailed study on C and carbon dioxide effluxes in thicket vegetation under different levels of degradation. This study thus aimed to determine the influence of thicket vegetation at various levels of degradation on soil carbon and carbon dioxide fluxes. The impact of plant photosynthetic pathway on soil C residence time and gas effluxes were analysed to elucidate on the land-use and cover patterns occurring in the area. All this was done to shed some light on the role of soil and thicket vegetation on carbon dioxide emissions and C storage in the spectrum of a shifting climate. The main area of research was Amakhala reserve in an Albany thicket in Eastern Cape Province; and it concentrated on three land cover types namely intact thicket, degraded thicket and grassland.

The objectives mentioned above were achieved by assessing historical rainfall variability from 1970 to 2010 through trend and time series analysis at nine rainfall stations located at Amakhala reserve, Grahamstown, Bathurst, Port Alfred, Uitenhage and Port Elizabeth. The land use changes that have occurred in the Albany thicket region covering Amakhala reserve, Grahamstown, Bathurst and Port Alfred were also assessed for 1989, 1999 and 2009 through satellite image analysis with Idrisi Andes GIS software; then their interaction with rainfall variability were determined. To elucidate on the vegetation species composition and land use / cover changes that have occurred in the study area, plant biomass as well carbon (C) and nitrogen (N) isotope measurements were done. Plant biomass was assessed for the dominant species through use of pre-existing allometric equations that required data on plant basal diameter, canopy area, stem numbers and height. The plant carbon was then estimated through use of a conversion factor of 0.48 on above-ground biomass, while soil organic C was determined through the modified Walkely - Black method. Carbon and N isotope ratios were determined from the foliar material of three replicate samples of dominant plant species then analyzed through mass spectrometry. Soil carbon dioxide effluxes were then monitored in each of the intact thicket (IT), degraded thicket (DT) and grassland (G) over a 10 month period; by measuring the net carbon dioxide exchange rate (NCER) through the dynamic chamber method. An automated carbon dioxide exchange analyzer, coupled to a soil temperature probe and photosynthetic active radiation (PAR) sensor was used; with NCER measurements taken every 20-30 days. Soil temperature, moisture, penetration resistance and PAR readings were taken during each assay and later used to interpret the NCER.

Results showed that long term variability in annual rainfall had a declining trend at Grahamstown ($r = -0.59$), Uitenhage and Bathurst stations ($r = -0.32$ at both stations), but was not significant at Amakhala, Port Alfred and Port Elizabeth stations. Most reductions in rainfall occurred in the 1980s and 1990s with the autumn, winter and summer rainfalls, the daily rainfall index and the daily rainfall subclasses of 10 mm and above showing a similar trend. The land use change detection gave a significant increase in proportion of degraded and transformed (moderately degraded) land between 1989 and 2009 with most of the increases occurring from 1989 to 1999, while farmland area decreased by 1.8 % over the years. Thus the Albany region had over 30 % of its land occupied by transformed vegetation, with heavy browsing and uncontrolled grazing being attributed to the destruction of pristine vegetation. Land-use change to game ranching and goat pastoralism was attributed to the reduction in farmland. Rainfall variability – land use change linkages were most significant in 1999 that recorded the least rainfall and had the lowest mean, maximum and sum of the NDVI. Grahamstown had the most significant rainfall-NDVI trends as it had the lowest NDVIs in 1999 when rainfall was lowest, the highest NDVI in 1989 when rainfall was highest and moderate NDVIs in 2009 when rainfall was moderate.

Vegetation at the IT was characterized by a dense thicket with diverse growth forms of canopy trees, woody shrubs, succulent shrubs and ephemerals which mostly had the C_3 type of pathway. This was in contrast with the IT soil isotopy that showed more positive C isotope ratios, indicating a switch between C_3 and CAM photosynthesis in original vegetation. Most of the canopy trees had disappeared in the DT to be replaced by herbs, shrubs and grasses. As such, there was a huge difference in isotope ratios between DT plants and soils with the plants having mostly C_3 metabolism while the soil showed a predominance of CAM plants in previous vegetation, indicating significant changes in land cover. The G site mostly

comprised the grasses *Themeda triandra* and *Panicum maximum* and a few herbs. It maintained a dominance of C_4 metabolism in both plants and soils showing very little change in species composition over the years. Because of the higher species diversity at IT, its soil organic C was quite high reaching levels of 3.4 % (i.e. 3.4 t C / ha) in the top 10 cm then decreasing with depth ($p < 0.001$); but was moderate at DT (1.1-1.3 %) and very low at G (≤ 0.5 % C) ($p < 0.001$). In the same manner above-ground biomass was highest at IT i.e. 330 000 kg/ha; but was only 22 000 kg/ha in DT and as low as 6 700 kg/ha in G vegetation. High biomass at IT was mostly attributed to the succulent shrub *Portulacaria afra* and the canopy trees *Euclea undulate*, *Rhus longispina* and *Schotia afra*. This above-ground biomass translated to biomass C amounts of 158 000 kg/ha at IT, 10 600 kg/ha at DT and 3 200 kg/ha at G. Thus the IT had the highest while G the least and DT moderate plant and soil C sequestration ability. In all, the conversion of IT to DT led to a net loss of 147 000 Kg of biomass C / ha and 12 000kg less organic C / ha of land.

Soil carbon dioxide effluxes were however variable between seasons as they were affected by differences in soil properties and seasonal weather patterns. High soil moisture levels (up to 16 % gravimetric moisture) resulted in reduced soil penetration resistance (1 to 4 Kg/cm^2) which raised effluxes at G and DT sites (up to 1.2 µmols m^{-2} sec^{-1}) in winter, while low moisture (~ 2 %) resulted in hard dry soil (~ 14 Kgm^{-2} penetration resistance) with suppressed CO_2 effluxes in spring (~ 0.2 µmols m^{-2} sec^{-1}) especially in DT and G soils. Rising temperature generally caused accelerated gas emissions but only when moisture was not limiting (as was the case in IT). Thus the high summer temperatures (up to 40^oC) gave lower effluxes especially in DT and G (< 1 $µm^{-2}sec^{-1}$) due to limited moisture supply (< 10 %); while the Autumn period that had very high temperature (up to 48 ^oC) and good moisture (up to 16 %) saw accelerated soil CO_2 emissions (averaging 2 µmols m^{-2} sec^{-1}) from all cover types. The high biomass and litter fall at IT served as ready substrate for soil respiration as long as moisture was not limiting and temperatures were favourable, while reduced cover at DT resulted in poor moisture conservation and creation of hard dry soils in spring and summer with reduced respiration. It was concluded that the DT had high CO_2 effluxes in winter and reduced emissions in summer; while the opposite was true for the IT. All the cover types had minimal CO_2 effluxes in spring and accelerated emissions in autumn. The grassland on the other hand was a fairly moderate source or sink of CO_2 in most seasons compared with the other two covers. It was observed that an environment of good moisture

and low-moderate temperatures (such as that in the winter) minimises effluxes while maintaining good plant productivity.

It was concluded that thicket vegetation is a good sink of carbon that should be preserved in its natural condition to optimize its carbon sequestration potential. All three land covers served as sources or sinks of CO_2 depending on soil and seasonal conditions. Thus high moisture and low penetration resistance generally increased effluxes of thicket ecosystems. The effect of increasing temperature on effluxes was only significant when moisture was not limiting. Conditions of good moisture and low-moderate temperatures gave reasonable amounts of effluxes while maintaining good plant productivity. Though the dry soil conditions significantly reduced effluxes in all land covers; they were not desirable since they decreased plant productivity and ultimately its C sequestration potential. Moreover, prolonged dry conditions only serve to exacerbate recovery of thicket plants as they increase mortality of canopy species in degraded and transformed areas in comparison with intact thicket.

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CHAPTER 1: GENERAL INTRODUCTION

1.1 Background

Concern over the increasing concentration of atmospheric greenhouse gases (GHG) has necessitated interest in their study globally. Solomon *et al.* (2007) reported an increase in atmospheric carbon dioxide (CO_2) among other GHGs, from a pre-industrial value of 280 to 379 ppm in 2005. Global carbon (C) sinks include the atmosphere, forest, soil, ocean and fossil fuels. Carbon stocks are constantly being exchanged between these reservoirs. Nature acts to balance the C cycle, but anthropogenic activities have caused some disturbances to the cycle. The use of fossil fuels and land-use change are primary sources of increased atmospheric CO_2 (Solomon *et al.*, 2007).

The terrestrial ecosystem serves as a good reservoir of excess greenhouse gases from the atmosphere, particularly CO_2. Emissions from terrestrial carbon are currently at about one-third of all global GHG emissions annually, and terrestrial carbon management represents roughly half the cost-effective mitigation available globally up to 2030 (Havemann, 2009). The role of soil as a source and sink of carbon has until recently been regarded as a minor one and overlooked by atmospheric scientists. Ontl and Schulte (2012) highlighted that soil stores up to four times more carbon than plant biomass and almost three times more than the atmosphere. It is essential to quantify this portion of C reserve, as a slight disturbance to it will affect the density of atmospheric GHGs leading to global climate change (Zhang *et al.*, 2008).

Soils may work as sinks or sources of carbon into the atmosphere, depending on the natural conditions and management type adopted. Land-use change and tillage cause major disturbances to the soil organic carbon reserve pool. In their study on the effect of long term tillage on C fluxes, Ussiri and Rattan (2009) observed that soils under no tillage stored twice as much C (about 813 kg $ha^{-1}year^{-1}$) than soil under conventional tillage. Conservation practices such as direct planting techniques, controlled grazing, planted forests and agroforestry systems can reduce C losses, by maintaining high levels of soil organic matter (Corsi *et al.*, 2012). Many aspects of the carbon cycle remain poorly understood. Part of the uncertainty is due to substantial spatial and temporal variation in terrestrial C pools and fluxes. Estimations of regional soil organic carbon (SOC) stores are sensitive to map scale effects, the geography of soil resources, and uncertainty stemming from organic carbon values assigned to soil series (Galbraith *et al.*, 2003). When measuring and modelling the C cycle at multiple scales (e.g. plot to region) the approach is to measure key carbon inputs, internal pools, and outputs.

South Africa has a rich diversity of vegetation. Mucina and Rutherford (2006) mapped nine biomes that include Fynbos, succulent Karoo, Desert, Nama-Karoo, Grassland, Savanna, Albany Thicket, the Indian Ocean Coastal Belt and Forest. The thicket biome in particular is a unique assemblage of spiny evergreen shrubs, succulents (trees, aloes and euphorbias), and a wealth of climbers with very little grass undergrowth (Knight and Cowling, 2006). It is mostly found where there is a degree of fire protection, but rainfall is too low. It cannot be grouped under Forest as it does not have the required height and the many strata below the canopy; nor is it a Savanna due to absence of a conspicuous grassy ground layer (Mucina and Rutherford, 2006). Thicket is most extensive in the southeast of the country, along the coastal parts of the Gouritz, Gamtoos, Sundays and Great Fish River valleys. It forms the western sector of the Maputaland – Pondoland - Albany biodiversity ‘hotspot’, which is defined by the high incidence of endemic succulents and bulbs (Knight and Cowling, 2006). Thicket Biome makes up 2.5% of the area of South Africa, i.e. nearly 31 500 km^2 (Pierce and Mader, 2006). The greater part of this biome has unfortunately been transformed or degraded through browsing by herbivores such as elephant and rhino, and over-grazing by livestock animals like goats (Pierce *et al*., 2008). Extensive research has been done on the potential of thicket biome to sequester C. Pierce *et al*. (2008) reported Spekboom (*Portulacaria afra*) thicket to store at least 20 kg Cm^{-2} of vegetation; making it effective at sequestering carbon from the atmosphere. Whereas considerable work has been done on the effect of converting forest or grassland to agricultural land on CO_2 emissions (Batjes and Bridges, 1992; Ussiri and Ratttan, 2009), little has been done on the consequence of this same disturbance on thicket biome. It is therefore essential to assess the influence of transformation of this biome on CO_2 emissions from the soil.

1.2 Problem statement

The thicket biome is quite extensive in the southeast parts of South Africa, in spite of the low rainfall and marginal soil conditions experienced in this area. Wide research has been done on its potential to sequester C, but very little has been done on its influence on soil CO_2 effluxes. It is evident from the changes in rainfall intensities, duration and amounts; increased temperatures, recurrent droughts, cyclones and fires that there are climate shifts in Southern African (Kiker, 2000). The concentration of GHGs in the atmosphere is constantly rising. Understanding the pools and fluxes of some of these gases is vital for developing viable strategies for mitigating climate change. The Eastern Cape Province is characterised by a mostly semi-arid environment with degraded land. It contains extensive areas of thicket vegetation which is affected by agricultural practices, industrial expansion, invasion of alien species and the over-grazing of domestic herbivores (Coastal and Environmental Services, 2003). Approximately 51% of the Province is largely open areas of natural vegetation that includes forest, woodland, grasslands, shrub-lands and low fynbos (CSIR, 2004). It is essential to study the extent to which vegetation at various levels of degradation can sequester excess CO_2 from the air. If C could be sequestered and durably stored in the soil, climate change could be reduced. It is also important to assess the influence of ^{13}C and ^{15}N isotope ratios as indicators of land-use change and their effect on soil organic matter turnover.

1.3 Aim of the study

The aim of this study was to assess the influence of thicket vegetation at various levels of degradation on soil carbon and CO_2 effluxes. This was done to elucidate on the role of soil and thicket ecosystems on C storage in the face of a shifting climate and changing land-use. Three land cover types namely an intact thicket, a degraded thicket and grassland were compared for purposes of this research.

1.4 Specific objectives

The specific objectives of the research were:

1. To assess the historic rainfall variability in an Albany thicket region and its relationship with temporal land-use and cover changes.
2. To quantify plant and soil C sequestered under thicket, grassland and degraded land.
3. To assess the influence of ^{13}C and ^{15}N isotope ratios as indicators of land-use change and their effect on soil organic matter turnover.
4. To investigate the impact of land-use type and soil properties on CO_2 effluxes from the soil.

1.5 Research questions

The research questions that needed to be investigated included:

1. What has been the trend in temporal land-use cover change and rainfall variability?
2. Can rainfall variability explain the land use cover change trends in the study area?
3. What is the importance of thicket biome and its influence on C storage?
4. Can C isotope ratios help infer land-use cover change and soil C residence time?
5. To what extent do thicket plants and soil act as sources and sinks of CO_2?
6. What impact do soil properties and the land use / cover type have on CO_2 effluxes?
7. Could rainfall variability and land-use cover changes in turn explain carbon flux conditions and carbon isotope composition of soil organic matter under different thicket vegetation conditions?

1.6 Thesis outline

This thesis comprises seven chapters. Chapter one gives a general introduction, the problem statement, aim and objectives of the study. A review of literature pertaining to carbon and carbon dioxide effluxes in different reservoirs is presented in Chapter 2. The chapter also elucidates on the use of carbon isotope ratios as a measure of land-use change and soil carbon residence time. In addition, a general overview of climate variability and land-use change that have been occurring over the years is done in this chapter. Chapters three to six zone in on the specific objectives of the study. Thus chapter three looks at the first objective which is to assess historic rainfall variability and its interaction with temporal land-use/cover changes. This provides a background to the quest for an understanding of the influence of vegetation changes on CO_2 effluxes; and ^{13}C and ^{15}N isotope ratios as indicators of land-use change and soil organic matter turnover. These details are then mapped out into respective chapters as follows: Chapter four addresses the second objective which is to quantify plant and soil C sequestered under different vegetation cover types; while chapter five focused on the third objective which is to assess the influence of ^{13}C and ^{15}N isotope ratios as indicators of land-use change and soil organic matter turnover. The impact of land-use type and soil properties on effluxes is reviewed in chapter six; and a synthesis of all the chapters is presented in chapter seven. The thesis concludes by making recommendations and suggestions for future research directions. A list of all the references used is provided at the end of the report.

CHAPTER 2: LITERATURE REVIEW

2.1 Greenhouse gases emissions and their abundance in the atmosphere

Greenhouse gases include water vapour, ozone (O_3), the trace gases carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) as well as synthetic chlorofluorocarbons (CFCs) which trap heat in the lower atmosphere (Wallington *et al*., 2004). Their composition in the atmosphere has changed significantly and rapidly since the beginning of large-scale industrialization in the 1750s (IPCC, 2001). This has resulted in the lower atmosphere (troposphere) having a carbon overload that traps heat and expands into the higher atmosphere (stratosphere) where ozone is mostly found; causing degradation of this gas that has a natural protective effect against harmful solar radiation (Wallington *et al*., 2004). Increase in atmospheric GHG concentration is mostly attributed to contributions from industrial emissions, fossil fuel burning, deforestation, land-use change and biomass burning (Bartjes and Bridges, 1992).

A study by Le Quéré *et al*. (2009) revealed that global carbon dioxide emissions from fossil fuel burning, cement production and land use change (mainly deforestation) were 27% higher in 2008 than in 1990. The atmospheric concentration of CO_2 reached 387 ppm in 2009, a figure far above its natural pre-industrial level by more than 105 ppm (Tripati *et al*., 2009). This concentration was said to be the highest in the last 800 000 years, and potentially the last 3 to 20 million years (Luthi *et al*., 2008; Tripati *et al*., 2009). Methane, another important GHG is mostly emitted from industries like paddy rice and ruminant farming, biomass burning, coal, gas and oil mining as well as natural reservoirs such as wetlands, permafrost and peatlands (Allison *et al*., 2009). Recent studies by Rigby *et al*., (2008) showed that the global concentration of methane in the atmosphere increased since 2007 to 1800 ppb after almost a decade of little change; with Northern Hemisphere emissions contributing more to this increase. Atmospheric N_2O is a source of stratospheric nitric oxide that is involved in ozone depletion (Peterson and Baringer, 2009). It is the third most abundant GHG after CO_2 and CH_4; and had a 1998 abundance of 314 ppb (IPCC, 2001), which rose to 319 ppb in 2005 (Le Treut, 2007) and is increasing at a rate of 0.5 – 0.9 ppb atmospheric concentration per year (Wallington *et al*., 2004). Agricultural activities, including the use of man-made fertilizers, industrial by-products, human and animal waste, and catalytic converters lead to higher nitrous oxide emissions. CFCs; mostly used in refrigeration, fire suppression systems and manufacturing processes are halogenated organic compounds in which all hydrogen atoms have been substituted by fluorine and chlorine atoms. Examples include CFC-12 (CCl_2F_2) and CFC-11 (CCl_3F), which are the two most abundant CFCs in the atmosphere present at levels of 0.5 and 0.25 ppb respectively (Wallington *et al.*, 2004). Studies have shown that a CFC molecule traps 20 000 times more heat than a CO_2 molecule, making it even more lethal in terms of its global warming potential (Esser, 1991; Le Treut, 2007). The concentration of synthetic CFCs increased until the 1990s, when the use of halocarbons was restricted under the Montreal protocol leading to a substantial decrease in their release (Le Treut, 2007).

2.1.1 Effects of GHG emissions on the climate

The climate is a complex interactive system comprising the atmosphere, land surface, snow and ice, oceans and other bodies of water and living things all powered by solar radiation (Le Treut, 2007). Net radiation (i.e. the difference between incoming solar radiation absorbed by the Earth-atmosphere interphase and the long wave radiation emitted from the Earth-atmosphere into space) determines the energy available for heating the atmosphere, ocean and land (Wallington *et al.*, 2004). Because of this, it has a bearing on seasonal rainfall variability and the strength of the global circulation patterns. Changes in atmospheric concentrations of GHGs and aerosols, land cover and solar radiation thus alter the energy balance of the climate system. Increase in concentration of atmospheric GHGs through anthropogenic activities alters the radiative balance of the Earth. This is because the gases absorb long wave radiation emitted by the Earth, but are transparent to incoming radiation from the sun, causing an increase in net radiation at the top of the atmosphere.

Reports by Allison *et al*. (2009) among others, state that temperatures have increased at a rate +of 0.19°C per decade over the past 25 years, a trend that is proportional to increases in anthropogenic greenhouse gas emissions over the years. They went on to suggest that twenty more years of emissions would give a 25% probability of warming to exceed 2°C, even with zero emissions after 2030. Allen and Soden (2008) observed a distinct link between extreme precipitation events and human induced warming through satellite observations and climate model simulations, with heavy precipitation events increasing during warm periods and decreasing during cold periods. Other consequences of global warming include accelerated melting of ice-sheets, glaciers and ice-caps in Greenland and Antarctic; rapid sea ice decline in the Arctic and sea level rise of 3.4 mm/year over the past 15 years. The IPCC Fourth

Assessment Report (IPCC, 2007) concluded that many changes in extreme weather events have occurred since the 1970s as part of warming of the climate system such as more frequent hot days, hot nights and heat waves; fewer cold days, cold nights and frosts; more frequent heavy precipitation events; more intense and longer droughts over wider areas; and an increase in intense tropical cyclone activity in the North Atlantic without a clear trend in total tropical cyclones numbers.

2.1.2 Climate variability in Africa

Africa's climate is controlled by complex maritime and terrestrial interactions that produce a range of climates across many regions, i.e., from the humid tropics to the hyper-arid Sahara (Boko *et al.*, 2007). The El Niño-Southern Oscillation (ENSO) is mostly responsible for inter-annual climate variability over eastern and southern Africa (Nicholson and Entekhapi, 1986) while some of it is under the influence of sea-surface temperature (SST) anomalies of the surrounding Indian and Atlantic oceans (Reason *et al.*, 2006). Eastern Africa is in sync with warm ENSO episodes, while southern Africa is negatively correlated; with dry conditions prevailing during warm ENSO events (Richard *et al.*, 2001). Increased inter-annual variability has been observed in this region post 1970, e.g. higher rainfall anomalies and more intense widespread droughts (Fauchereau *et al.*, 2003). Intense droughts have been recorded in Zambia, Malawi, Zimbabwe, and northern South Africa while Angola, Namibia, Mozambique, Malawi, Zambia have experienced a significant increase in heavy rainfall events (Reason *et al.*, 2006; Usman and Reason, 2004). There is further evidence for changes in seasonality and weather extremes (Washington and Preston, 2006).

Across West Africa, large-scale changes of SSTs in the Atlantic, Pacific, Indian Ocean, and Mediterranean Sea promote changes in major atmospheric circulations that influence year-to-year seasonal climatic variations (Christensen *et al.*, 2007). The wet years of the Sahel for example are characterized by warm Atlantic SSTs compared to dry years, while in the dry years, the African Easterly Jet is generally stronger than in the wet years and is displaced towards the equator, and the Tropical Easterly Jet stream is unusually weak (Nicholson, 2001). The Inter-tropical Convergence Zone (ITCZ) is often displaced further south and / or is abnormally weak in dry years. ENSO also influences climate variations (but not in a dominant way) with El Nino episodes giving negative rainfall anomalies over the Sahel and some regions of West Africa (Wilby, 2008). In West Africa, major declines in annual rainfall have been observed between the periods 1931-1960 and 1968-1990 (Dai *et al*., 2004). Nicholson, (2001) noted a significant long-term reduction in rainfall in the semi-arid regions of West Africa by 20 to 40% in parts of the Sahel; with anthropogenic activities, particularly land use change and desertification, being proposed to have caused this decline.

The African Easterly Jet (AEJ) and Tropical Easterly Jet (TEJ) are important for the development of the summer rainfall over northern Africa (Nicholson, 2000). The North Atlantic Oscillation (NAO) also affects inter-annual climate variability, particularly winter temperatures in this region; with drier-than-normal conditions occurring during high NAO index winters (Marshall et al., 2001). Anomalous temperature variations (cooling) over North Africa and the Middle East, associated with the stronger clockwise flow around the subtropical Atlantic high-pressure centre are notable during high-index NAO winters (Hurrell *et al*., 2000). Generally almost all of Africa has been affected by increased aridity, especially since the 1980s (Nicholson, 2001). The climate of Africa has also experienced extreme wet and dry intervals during the past two centuries, with warming in the 20th century at a rate of about 0.05°C per decade (IPCC, 2001).

2.1.3 Vulnerability of Africa to climate change

As a continent, Africa is vulnerable to climate variability and change as the livelihoods of the people are often directly linked to the climate. According to IPCC (2001) key sectors that are vulnerable to the manifestations of climate change include:

1. *The water resources sector*: especially international shared basins where conflicts might arise and there is a need for coordinated water management.

2. *Food security*: is at risk from declines in agricultural production caused by an uncertain climate.

3. *Natural resources productivity*: biodiversity might be irreversibly lost in a changing climate.

4. *Health sector*: vector and water-borne diseases will arise in areas with inadequate health infrastructure and sanitation.

5. *Coastal zones*: are vulnerable to sea-level rise affecting roads, bridges, buildings, and other infrastructure that is exposed to flooding.

6. *Desertification*: will be exacerbated by changes in rainfall and intensified land use.

The low adaptive capacity of the continent further compounds this vulnerability. It is envisioned that by 2020, a large proportion of the population would be exposed to increased water stress due to climate change induced water unavailability, coupled with increased water demand (Davies, 2010). Yields from rain-fed agriculture will be substantially reduced in some areas, further threatening food security and exacerbating malnutrition.

CO_2 is the most abundant GHG, thus more emphasis was placed on it in this study. The vulnerability of Africa to the consequences of climate change (e.g. reduced water resources and declining rain-fed agriculture productivity) necessitates sustainable landscape management options that promote optimal land-use measures which integrate production, environmental protection, climate change adaptive and mitigation strategies as well as carbon sequestration initiatives. Carbon sequestration options require an understanding of C sources and sinks as well as processes governing their effluxes in these reservoirs.

2.2 Sinks of carbon

Carbon is the basis of life on Earth. It is incorporated into plants through photosynthesis, absorbed by animals through their food, present in the atmosphere as carbon dioxide, locked into rock as limestone, and pressed into fossil fuels such as coal and oil, (Houghton, 2002). It is therefore essential to study the quantity and fluxes of carbon in different pools. The C cycle is a biogeochemical cycle by which C is exchanged between the terrestrial biosphere (freshwater systems and non-living organic material constituting soil C), geosphere, hydrosphere and atmosphere of the earth (Lal, 2008). It is made up of four major reservoirs of C interconnected by pathways of exchange. The global C budget is a balance of exchanges of C between these reservoirs.

Mitigation options for limiting the increase of GHG concentration in the atmosphere comprise either emission reduction or removal and storage of greenhouse gases. Carbon sequestration i.e., the capture and secure storage of C than would otherwise be emitted to or remain in the atmosphere, (Cole, 1995) acts as a good option to address the problem of excess carbon. The aim is to capture C emissions produced by human activities and divert them to secure storage. Carbon sequestration activities do not avoid the production of CO_2, but lock it away from the atmosphere for a certain period of time (Jung, 2005). This long-term storage of carbon can take place in the oceans, forests and soils. The quantities and composition of atmospheric C has already been alluded to in section 2.1, we now concentrate on the other reservoirs.

2.2.1 C in the hydrosphere and geosphere

The ocean contains about 36000 gigatonnes of C, mostly in the bicarbonate form (Houghton, 2005). C exchange is important in controlling oceanic pH. Regions of oceanic upwelling cause C release into the atmosphere, while down-welling transfers CO_2 from the atmosphere to the ocean. Two strategies for enhancing carbon sequestration in the ocean include the enhancement of the net oceanic uptake from the atmosphere by fertilization of phytoplankton with micro- or macronutrients, and direct injection of relatively pure CO_2 stream to ocean depths greater than 1000 meters (Lal, 2008). Sources of CO_2 for direct injection might include power plants and industries. In the geosphere, C exists in rocks and sediments or fossils (Houghton, 2005). Artificial sequestration of C into the geosphere involves capture, liquefaction, transport and injection of industrial CO_2 into deep geological strata (Lal, 2008). The CO_2 may be injected in coal seams, old oil wells, stable rock strata or saline aquifers.

2.2.2 C in the terrestrial biosphere

Around 1900 gigatonnes of C are present in the biosphere where it plays an important role in the structure, biochemistry and nutrition of living cells (Wikipedia). In the biosphere it occurs in plants, soils and microorganisms. Processes such as respiration, decomposition, burning, afforestation and reforestation affect biosphere C cycle, (Janzen, 2004). The long-term conversion of grassland and forestland into cropland has resulted in significant disturbances to the terrestrial C sinks (Cerri *et al.*, 2007). Conservation practices such as afforestation / reforestation and conservation farming help restore degraded land through reduction of soil erosion and increases in soil organic matter (Lal, 1997). In the past, opinions varied as to whether C sequestration in soils would be realistic and practical. In recent years, positive evidence has shown this to be possible. It has been reported that it is possible to remove between 40 and 80 billion tonnes of C between the next 50 to 100 years by sequestering it into cropland soils (Cole, 1995). Improved management of the carbon stored in terrestrial vegetation and soil is therefore a necessary effort to avoid dangerous climate change. Above and below ground terrestrial carbon pools are vital carbon stores, and are significant environmental assets, but are potential environmental liabilities if threatened. Soil C sequestration might offset the adverse effects of fossil fuel burning and land-use change. Carbon sequestration in agricultural land also counteracts desertification through increased soil organic matter and structural stability, increased water retention and prevention of soil erosion.

2.3 Measuring and monitoring terrestrial C

Carbon pools and fluxes require quantification to fully understand their cycling processes and potential impacts of climate change. The terrestrial ecosystem stores a large proportion of C, so it is essential to quantify this portion. Terrestrial C can be measured in various pools such as above and below ground biomass, soil organic matter, litter, dead wood and harvested wood products. The above-ground biomass pool includes all biomass of vegetation both woody and herbaceous above the soil like stumps, branches, bark, seeds and foliage (IPCC, 2006). The below ground biomass pool comprises all live roots, excluding all fine roots less than 2 mm diameter since they often cannot be distinguished from SOM. Dead wood consists of all non-living woody biomass not contained in litter, either standing, lying on the ground or in the soil. This includes tree stumps and dead roots at least 10 cm in diameter. Methods for measuring pool carbon include soil and vegetation inventory, C flux measurements, satellite remote sensing, ecosystem modeling and biome models (Watson *et al*. 2001). The choice of method depends on measurement scale, costs, capacity requirements (e.g. equipment and technical know-how), precision, accuracy and the frequency with which the method can be applied (whether for initial stock measurement or periodic monitoring).

2.3.1 Above and below ground biomass measurement

This can be done through destructive harvesting of the tree to determine biomass through actual weight of components like stem, branches and foliage (Havemann, 2009). It is an accurate method within a small unit area but is expensive, time consuming, damaging to the environment and infeasible at large scale. This method is mostly used to calibrate allometric equations. One can also make use of non-destructive allometric methods by conducting a field inventory of specific factors e.g. species, stem density, diameter at breast height (DBH) and height. An appropriate equation/model is then used to convert the measurements into biomass estimates. Common conversion equations include dry wood density conversion to wood dry weight; biomass conversion factor; e.g. root: shoot ratio; biomass conversion to C content and fresh biomass conversion to dry biomass (Watson *et al*. 2001). Non-biomass C pools e.g. dead wood, litter and harvested wood products are estimated using known relationships with above-ground biomass such as leaf area index (LAI), net primary productivity (NPP), crop yield and litter cover (IPCC, 2006).

2.3.2 Remote sensing

This captures the spectral and spatial characteristics of an area through use of aerial photographs or satellite imagery and is therefore efficient to estimate vegetation cover, density and structure (Petrokofsky *et al*., 2012). Multi-spectral satellite remote sensing is used to detect variations in the leafiness of the land cover from its reflectance by use of an index known as the normalized difference vegetation index (NDVI), which is based on the reflectance at two wavebands in the red and near infrared parts of the solar spectrum (Grace *et al*., 2007). The NDVI is essentially a measure of ‘greenness’ and can be used to estimate the leaf area per unit of land area i.e. LAI. Remote sensing has been used to record land-use and cover change and is suited to capture large-scale deforestation events. It can collect information in inaccessible areas and may allow for repeated coverage (Goetz and Dubayah, 2011). Measurement of smaller-scale events e.g. degradation or intensification of agricultural changes requires more detailed data and interpretation. The major limitation with remote sensing is that continuity of sensor types across a suite of spectral, temporal, and spatial scales is not assured (e.g. for Landsat). Estimation of soil organic matter (SOM) using remote sensing relies on the strong relationship between SOM quantity and soil colour (visible reflectance) i.e. organic matter is dark coloured; so this requires visibility of bare ground, good calibration and ground truthing. The limitation in use of soil reflectance is that it is a function of soil moisture, texture, chemical composition, parent material and surface conditions, (soil moisture like organic carbon, decreases reflectance especially within the visible bands) (Gomez *et al*., 2008).These can vary at any time and are further magnified when a large geographic area has to be mapped. Again remote sensing has good correlation in areas with moderate to high SOC levels (>2.0%), but is not effective in soils with low SOC due to the poor reflectance of such areas (Stephens *et al*., year unknown).

2.3.3 Soil C determination

It is essential to quantify the soil C reserve as a slight disturbance to it will affect the density of greenhouse gases in the atmosphere leading to global climate change (Zhang *et al*., 2008).

Traditional methods used include the loss on ignition method which is SOM measurement of sample weight change after oven-drying (Brady and Weil, 2002). This can over-estimate OM due to supplementary weight loss of inorganic constituents, mostly hydrated clays during heating; as a result, it is affected by ignition temperature and sample size (Gehl and Rice, 2007**)**. The Walkley-Black acid digestion uses chromic acid to measure oxidizable soil organic C. It is however inaccurate for soils with very high and very stable C e.g. peat soils (Brady and Weil, 2002). These methods are further complicated by the spatial and temporal variability inherent in soil horizonation, texture and bulk density. Spectroscopy is one in-situ soil C determination method that is non-destructive hence minimizes soil disturbance while increasing the ability to analyse large areas. Here, middle or near-infrared reflectance spectroscopy is used to measure soil organic C in conjunction with dry-combustion analysis (Gehl and Rice, 2007**)**. It is less costly than traditional methods and greatly increases the speed of analysis. The use of portable field remote sensing tools for the determination of in-situ soil C may be effective in estimating soil C constituents that require expensive and time-consuming analytical methods e.g., litter decomposition rates, microbial biomass and microbial activity.

2.3.4 C isotope measurements

Natural stable isotope ratio determination is a technique used to find out about the geographic, chemical, and biological origins of material (Muccio and Jackson, 2009). It can be used to determine the source of an organic substance based on the relative isotopic abundances of the elements comprising it. Plants contain less ^{13}C than the atmosphere, as they discriminate against $^{13}CO_2$ during photosynthesis due to the biochemical properties of their carbon fixing enzymes and limitations to CO_2 diffusion into the leaf (Boutton *et al*., 1998). This ^{13}C depletion varies profoundly and consistently among plants using different photosynthetic pathways i.e. the Calvin cycle (C_3), Hatch–Slack cycle (C_4) and Crassulacean acid metabolism (CAM) photosynthetic pathways; that ecologists have used it to distinguish between plant species in large-scale surveys (Michener and Lajtha, 2007). Within the leaf, the carboxylation enzyme ribulose bisphosphate carboxylase (prominent in C_3 plants) further discriminates against ^{13}C more than phosphoenol pyruvate carboxylase (prominent in C_4 plants) (Boutton *et al*., 1998). Because of this, C_3 plants have less ^{13}C than C_4 plants. Stable isotope ratios of plants and soil integrate information relating to the identity of the processes that formed them, the rates of those processes, and the environmental conditions prevailing at the time. They are used in ecological studies as tracers to investigate structural and functional characteristics of ecosystems and their responses to environmental changes and human activities (Bai *et al*., 2012). Plants with C_3 photosynthesis have $\delta\,^{13}C$ values ranging from approximately -32 to -22‰, while C_4 plants range from about -17 to -9‰. The obligate CAM plants have δ 13C values similar to C_4 plants; but facultative CAM plants range from -30 to -10‰, depending on the relative amount of carbon fixed by CAM versus C_3 photosynthesis (Boutton *et al*., 1998). ^{13}C analysis has thus become an accepted method for determining the pathway of CO_2 fixation of terrestrial plants.

In natural ecosystems, soil organic carbon is derived predominantly from the residues of native vegetation; hence the stable carbon isotope ratio of the vegetation has a direct impact on the $\delta\,^{13}C$ of the soil organic matter (Kao, 1997). In areas where vegetation has changed from one photosynthetic pathway to another, the isotopic signature from the original vegetation persists in the soil depending on the turnover rate of SOC in that ecosystem thereby revealing the combination of inputs from both past and current vegetation (Bai *et al*., 2012). The characteristic $\delta\,^{13}C$ of vegetation thus serves as a marker for the origin of soil organic matter (Michener and Lajtha, 2007) after land conversion.

2.4 CO_2 effluxes in soil and their measurement

Since soils and their biota play a significant role as sources and sinks of GHGs; it is important to note what proportion of the GHGs entering the atmosphere are derived from soil, how the amounts evolved are affected by land-use and management; and to what extent soils act as a sink for these gases.

2.4.1 Mechanisms of gas transport in the soil

Gaseous transport in the soil is effected by convection (or mass flow) and diffusion; with the former induced by total gas pressure between the soil and atmosphere; while the latter is driven by a partial pressure gradient of any constituent member of the gas mixture to move from areas of higher concentration to that of lower concentration (Lal and Shukla, 2005). Convection is induced by atmospheric pressure changes, temperature gradients and wind gusts (Hillel, 2003); and is also affected by gaseous permeability, which is in turn dependent on total porosity, pore size distribution, and tortuosity of continuous air-filled pore space (Nkongolo *et al*, 2010). Diffusive gas transport on the other hand depends on the total volume and the tortuosity of continuous air-filled pore space. The pore tortuosity factor is an important consideration since it defines the pathway of gas diffusion, affected by pore geometry thereby giving rise to differences in diffusion rates between poor and well-structured soil (Bartjes and Bridges, 1992). Diffusion has been seen as the most significant process in soil gas transport (Nkongolo *et al*., 2010).

2.4.2 Soil respiration

The concentration of CO_2 in the soil is 10 or more times greater than that in the atmosphere (Hillel, 2003). This mostly comes from respiration of plant roots, soil macro and microorganisms. The contribution of soil fauna to respiration is however much less than that of microbes; while root respiration accounts for 30-70 % of total soil respiration (Bartjes and Bridges, 1992). Factors affecting soil respiration include soil properties such as temperature, wetness, pH, and organic matter content and composition; all of which influence the time-variable respiratory activity of soil organisms (Hillel, 2003).

2.4.2.1 Soil thermal properties, moisture and season effects on soil respiration

Soil thermal properties not only look at temperature but include such properties as thermal diffusivity, conductivity and resistivity. Nkongolo (2010) found significant correlations between CO_2 effluxes and soil thermal diffusivity, conductivity and resistivity where temperature could not account for this variation. Both temperature and moisture usually increases respiration since they have a positive effect on organic matter mineralization and microbial activity (Schaefer *et al.*, 2009). Microbial respiration more than doubles for every 10°C rise in soil temperature up to a maximum of about 40°C, beyond which temperature becomes too high, limiting plant growth, microbial activity and soil respiration (USDA, 2009). On the other hand; soil respiration increases with increasing soil moisture up to the level where pores are filled with too much water limiting oxygen availability, which interferes with the soil organism's ability to respire. Thus in dry soils, respiration declines because the soil moisture deficit limits microbial activity; while the low oxygen levels of extremely wet soils result in poor organic matter decomposition (Yuste *et al*., 2007). Xie *et al.* (2010) found variations in seasonal soil respiration among different vegetation types, with soil water content and temperature being the main influences. Moreover, soil respiration was affected by soil organic matter and total nitrogen when the water and heat condition were plentiful, but the two had a negative impact on soil respiration when seasonal water and heat were below optimal. Wang *et al.* (2007) noted that microbial respiration depended on temperature, moisture, substrate quality and quantity, maximum activity of respiratory enzymes and demand for respiratory products with microbial respiration being higher in summer and lower in spring and autumn, which corresponded with changes in soil temperature.

2.4.2.2 Litter fall, land-use pattern and management effect on soil respiration

The activities of soil microorganisms depend on the input of organic matter from above ground and root biomass (Fang *et al.*, 1998). This is because litter fall affects soil organic matter content, pH, temperature and moisture; hence soil respiration varies remarkably with vegetation cover. Schaefer *et al.*, (2009) reported above-ground plant litter fluxes to be strongly correlated with soil respiration with litter fall removal reducing respiration. They explained that above-ground biomass, belowground root turnover and exudation provided organic matter to the soil thereby increasing respiration. In general, plant residues with a low C: N ratio decompose faster than those with high ratios since nitrogen limits enzyme production, microbial biomass and ultimately SOM decomposition (Yuste *et al.*, 2007).

Different land-use patterns e.g. cultivation and irrigation not only change the vegetation cover but properties of the soil (e. g. porosity, drainage, organic matter quality and amount, composition and activity of microbes) and consequently soil respiration (Fang *et al.*, 1998). Soil management practices that affect SOM, moisture, aggregation and pH will influence soil respiration. Thus tillage, residue burning and removal diminish SOM content and microbial activity by reducing aggregate stability and porosity thereby reducing respiration (USDA, 2009).

2.4.3 Measurement of soil respiration

Methods for measuring GHG effluxes include the use of chambers, soil core incubators and eddy-covariance towers. Chambers involve placing an open-bottom chamber over a small area of soil surface and measuring the gas emitted into it (Davidson, *et al*., 2002). The trapping may be passive (static, closed) where the gas in a closed chamber is sampled periodically, and efflux computed from the concentration increase inside the chamber e.g. use of PVC chambers; or active (dynamic, flowing) as in the use of infra-red gas analysers (Bekku *et al*., 1997). Chambers are simple and easy to operate but if not well designed could change the environmental properties such as temperature and the wind profile near the soil surface; and consequently the gas flux pattern (Davidson, *et al*., 2002). With incubators, soil cores are homogenised and submerged in water then incubated in situ for months before gas emission can be measured. Emission is estimated by headspace sampling over a period of time, an example would be the use of the alkaline trap soda lime (Cheng and Coleman, 1989). While cores are cheap and allow for collection of data with good spatial resolution; they are limited in that homogenisation and incubation processes can affect emission rates so field calibration may be necessary (Cable *et al.*, 2012). Soil incubation cores are adequate for studying production or oxidation processes occurring in soils or sediments but cannot be used for flux determination. Eddy-covariance towers on the other hand are a technique that measures and calculates vertical turbulent fluxes within atmospheric boundary layers (Baldocchi, 2003). It is a statistical method used in meteorology and other applications that analyse high-frequency wind and scalar atmospheric data series, and yields values of fluxes of these properties. Its advantage is that it has a large spatial coverage and is well suited for terrestrial systems **(**Nagy *et al.*, 2011). However, it is extremely expensive, since construction of high towers is needed, the technique is mathematically complex, and requires care in setting up and processing data. Again it is wind speed and direction dependant and gap filling is often needed to obtain annual seasonal budgets. In this present research, the dynamic chamber method was used to measure CO_2 effluxes from the soil.

The above was a general outline of the literature pertaining to soil C research. More specific literature pertaining to each objective is built within the relevant chapter, which are structured in publication format.

CHAPTER 3: THICKET VEGETATION RESPONSE TO HISTORIC RAINFALL VARIABILITY

This chapter focuses on assessing historic rainfall variability and its interaction with temporal land-use and cover changes. This was done to provide some background information on the general rainfall trends as well as land-use / cover changes that have been occurring in the Albany region in which the study was carried out. Such information would be useful later when quantifying the actual amounts of carbon and soil carbon dioxide effluxes occurring under different land cover types, and how these are affected by weather extremities. The land-use change detection also links up well with the chapter on assessing the use of isotope ratios as indicators of land-use change and soil carbon residence time. The chapter began by analysing historical rainfall variability at nine rainfall stations within the Eastern Cape Province through trend and time series analysis. Extreme rainfall events were also studied to get a clearer picture of the precise variations in rainfall that has been occurring. The chapter then assessed temporal land use and cover changes within the Albany thicket region in which the rainfall stations were located through change detection analysis. Further efforts were made to identify if there was any linkages between rainfall variations and land-use / cover changes in the study area.

3.1 Introduction

South Africa, like many parts of southern Africa, experiences limited water availability which is often of low quality (Boko *et al*., 2007; Dennis and Dennis, 2012). It is essentially an arid to semi-arid water-stressed country, (Davies *et al*, 1993; Dennis and Dennis, 2012). According to Walmsley *et al*. (1999), South Africa's available freshwater resources are almost fully utilized and under serious threat. Several factors contribute to this and these include; a natural climate characterized by low rainfall and high evaporation rates, a rapid population growth, expanding urbanization and increased economic development (Sharma *et al*., 1996 and DWAF, 2005). Low rainfall and high evaporation rates create low available stream run-off, while rapid population growth and economic development lead to greater water demand and increased pollution of available resources (Walmsley *et al*., 1999). Land use and cover change and its associated problems (e.g. erosion and siltation) also contribute to ecological changes on the hydrological cycle (World Bank, 1995). Public service delivery is sometimes hampered by a poor policy environment in some sectors, which is insufficient to deal with environmental degradation and disaster risks (Boko *et al.*, 2007). The policy pertaining to national management of water resources and land-use practices of a country ultimately impact water quality and availability.

South Africa receives mean annual precipitation of about 500 mm, with only 9% of this being converted to river runoff (Shippey *et al.*, 2004; Midgley *et al.*, 2007). This rainfall displays a decreasing trend from east to west, is highly variable within and between years and is punctuated by recurrent droughts (DWAF, 2008 and Davies, 2010). The consequences of limited freshwater supplies are that most of the country's major rivers have been dammed to supply industry, agriculture, and domestic needs (Davies *et al*, 1993). In addition, more than 50% of the wetlands have been utilised for agricultural and other purposes (Walmsley *et al.*, 1999). These changes have affected the habitat and biotic diversity of freshwater systems. The rapid rate of rural to urban migration and accompanying mushrooming of informal settlements has affected the provision of clean water and sanitation services in the cities (DWAF, 2005). Furthermore, industrial and domestic effluents are polluting the ground and surface waters, (Walmsley *et al*. 1999). Because of these, water shortages are predicted in the short- medium term in most of South Africa’s major towns of Rustenburg, Gauteng, Cape Town, Ethekwini, Nelson Mandela Bay, Polokwane and Lephalale (DWAF, 2009).

According to Blignaut *et al*. (2009), South Africa has been approximately 2% hotter and at least 6% drier between 1997 and 2006 compared to the 1970s. Rainfall trends are more difficult to determine, with significant regional differences evident in the whole of southern Africa (Boko *et al.*, 2007). More climate changes are expected to further affect the country’s hydrological systems and water resources (Schulze, 2005). According to DEA (2010), scenarios of possible temperature increases in the order of 1-3°C are predicted, and rainfall patterns are likely to become lower in the already semi-arid and arid West (with a 5-10% decrease in rainfall) and higher in the East (Dennis and Dennis, 2012). Dennis and Dennis, (2012) also postulated longer dry periods interspersed with more intense rainfall events associated with droughts, floods and decreased river flows in the country. It is against this backdrop that trends in climate patterns should be consistently analyzed.

Most of the Eastern Cape (EC) Province of South Africa has a bi-modal type of rainfall, thus it receives both summer and winter rainfall (DEA, 2010)**.** The climate is arid to semi-arid, receiving 350 - 550 mm rainfall per annum (Palmer, 2004). The region has a mean annual temperature of 17.6°C (mean monthly range of 12.3–22.4°C) with daily maximum temperatures in summer regularly reaching 40°C (Mills *et al*. 2005). The EC sits in a transition zone of topographical, geological, pedological and climatic complexity which give rise to a wide range of habitats that make it botanically diverse, (Kerley *et al*., 2003). Thus it comprises many biomes such as Albany thicket, Forest, Fynbos, Grassland, Succulent Karoo and Savanna (DEDEA, 2009). The Albany thicket has been identified as one of the subcontinent's eight biodiversity hot spots since it has approximately 2000 species with about 10% endemism (Kerley *et al*., 2003). This biome is rich in woody and succulent thicket species underlain by a Karoo rock sequence **(**Palmer, 2004**)**. It is also dominated by thorny, spinescent, often succulent bush known as subtropical thicket (Berliner and Desmet, 2007).

Questions have emerged as to what would be the impact of reduced rainfall and a higher incidence of drought on the biodiversity of these biomes. Petersen and Holness (Year unknown) postulated that there is likely to be shrinkage, shifts and increased extinction of major biomes, particularly endemics, with the most severe impacts predicted for the South and West winter rainfall regions of the Fynbos and succulent Karoo biomes. The long-term variation in rainfall, including years of both drought and high rainfall are all likely to cause shifts in plant biomass and species composition (Dean *et al*., 1995). Other threats to the proliferation of Albany thicket in EC include destruction by herbivores, clearing for cultivation, harvesting of medicinal plants and invasion by alien species (Palmer, 2004). This vegetation is particularly vulnerable to degradation. Over the years, cultivation in thicket has been reduced with most of the rangelands currently devoted to goat farming in the rural area and game farming in the commercial areas. Smith and Wilson (2002) reported a shift in land use in the commercial farms from pastoralism to game farming since the 1980s to be a fast growing trend in South Africa. On the other hand, Kerley *et al*. (2003) reported the dominant land use particularly in the rural area of Thicket biome to be now pastoralism of goats (*Capra hircus*); which has led to a large loss of the indigenous fauna and a collapse of ecosystem health. Dean *et al*. (1995) indicated that the fall in stocking rates by about 50% over 100 years imply a progressive loss of usable primary productive land and the degradation of rangelands. It is not clear what has been the effect of these changes in land use and rangeland degradation on the climate of the region and vice -versa. Considering the low recovery rate of thicket species once destroyed, it is imperative to assess the effect of rainfall extremes on this biome.

In light of this, the aim of the present study was (1) to assess historic rainfall variability within the Albany thicket region of EC South Africa; and (2) to understand the interactions between rainfall trends and land-use / cover changes over the years. Extreme rainfall events were also examined by analyzing daily frequencies of extreme wet and dry days over the years.

3.2 The study area

The EC Province spans the coastal area that lies between the subtropical conditions of KwaZulu Natal and the Mediterranean conditions of the Western Cape Province (Figure 3.1). Its inland is bisected by the great escarpment with a series of rivers and wetlands, while its northern areas comprise the Plateau and great Karoo (Palmer, 2004). These topographical differences give rise to the vast climatic differences experienced in its towns and cities. Not only does the climate vary according to proximity to the ocean, but it also gets progressively wetter eastwards. The EC incorporates aspects of both winter and summer rainfall (DEDEA, 2009). As a result, along the coast the climate is mild warm temperate to sub-tropical, with winter rainfall in the Southern Cape city of Port Elizabeth (DEDEA, 2009). It also comprises a warm coastal belt between Port Elizabeth and East London, a humid zone beyond East London with the climate becoming sub-tropical beyond Port St Johns. Rainfall in the region is poor, with droughts of several months common inland. It also experiences hot temperatures during summer (exceeding 40^oC) and close to freezing in winter (Palmer, 2004). The harsh climate gives rise to thicket plants with a high degree of succulence and leaf sclerophyll.

The present study was based at the Amakhala Game Reserve and its immediate cities of Port Elizabeth and Grahamstown, and the towns Port Alfred, Uitenhage and Bathurst (Figure 3.1). These areas were chosen since they comprise a lot of the Albany thicket vegetation and would help in identifying the vegetation cover changes that have occurred over the years in comparison with historical rainfall trends. A total of nine stations were assessed and these include one station each at Amakhala reserve, Port Elizabeth and Bathurst then two stations each at Grahamstown, Port Alfred and Uitenhage.

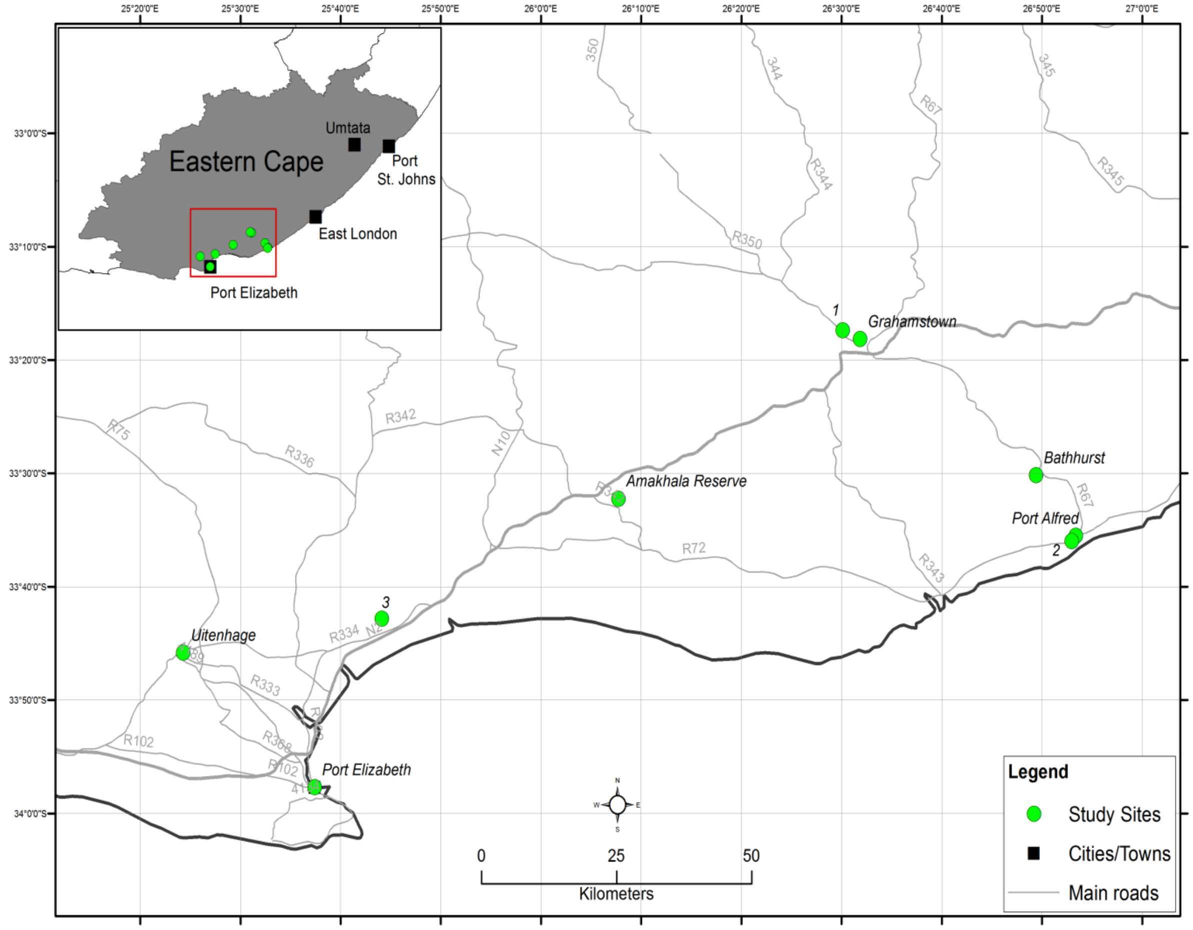


Figure 3.1: Map showing stations used for rainfall variability studies

3.3 Materials and methods

Rainfall data from 1970 to 2010 for nine stations was collected from the South Africa Weather Services (SAWS) and the Agricultural Research Council of South Africa (ARC); and analysed for historical trends using time series and trend analyses. The Bathurst station however only had daily rainfall data for 1970 to 2001.

3.3.1 Annual rainfall patterns

Annual rainfall for each station was analysed over the study period. It was standardized through use of the standardized anomaly index (SAI) denoted Z score:

$$\text{SAI or Z score} = (X - \mu) / \sigma$$

Where:

Z score = Standardized anomaly index

X = Annual rainfall for that particular year
μ = Mean annual rainfall for all years under study
σ = Standard deviation of annual rainfall for all years under study.

The Z score was assigned for each year per station. It shows clear trends in annual rainfall patterns. A Z score of zero means that the annual rainfall for that particular year is equal to the mean rainfall value of all years under study; while a positive value showed above normal and a negative score showed below normal rainfall.

3.3.2 Extreme rainfall events

Analysis of daily rainfall was done to identify the magnitude and frequency of extreme precipitation events. These were assessed for Grahamstown, Port Alfred, Uitenhage and Barthust stations since they were the only stations with daily data. The daily rainfall indices that were examined included the following:

3.3.2.1 Daily Rainfall Index (DI)

DI = Annual rainfall / Total number of wet days in the year

3.3.2.2 Highest daily rainfall for each year

This was done by scanning the daily records for each station and year and determining the largest 1 day event.

3.3.2.3 Frequency of dry spells, determined through observing:

a) Total number of days with no rainfall per year and
b) The largest (i.e. in terms of length) dry spell for each year.

3.3.2.4 Frequency of wet spells (or total number of wet days / year) in the daily rainfall class ranges of:

a) 0.1-9.9 mm;
b) 10-19.9 mm and

c) > 19.9 mm for each year.

However, it was not possible to analyse extreme rainfall events for Amakhala Reserve and Port Elizabeth Stations since they lacked daily data.

3.3.3 Statistical Analyses of rainfall variability

Data was analysed for variability using the Statistica software, Version 11 (Hill and Lewicki, 2007) through a time series analysis of all annual rainfall for each station over the entire study period. The Z- score was also subjected to linear trend and linear regression analysis (R^2 value at $p < 0.05$) using MS Excel. The degree of correlation between the above measured variables and time series was assessed using Pearson's product moment correlation coefficient (r value) at $p < 0.05$ significance level.

3.3.4 Seasonal rainfall patterns

To identify seasonal variations, monthly rainfall patterns for the entire study period were assessed per station through use of trend analysis. Months were then grouped into seasons based on the temporal rainfall patterns observed in each month.

3.3.5 Station Comparisons

Comparisons between stations were made by looking at the following parameters:

(i) Mean, maximum and minimum annual rainfall
(ii) Annual rainfall variability

3.3.6 Land use or cover change detection

Land use and cover (LUC) as well as their changes were investigated through satellite image analysis of the Albany thicket region of EC province using the Idrisi Andes GIS software. The images covered the area occupied by the two Grahamstown, two Port Alfred as well as the Bathurst and Amakhala reserve rainfall stations. Thus the Port Elizabeth and Uitenhage stations were not included in the LUC change analysis since they fell outside the image boundary. To achieve this, three Landsat images of the area obtained from USGS Earth

Resources Observation Systems Data Centre for the years 1989, 1999 and 2009 were used. There were no images for the 1970s period so these three years were selected to get an idea of the temporal variations in LUC over each decade. The images were geo-referenced to the Universal Transverse Mercator-UTM system then Normalized Difference Vegetation Index (NDVI) images were created for the different years from the near infrared (NIR) and red bands of Landsat TM image. The NDVI is defined as:

$$NDVI= (NIR-R) / (NIR+R)$$

Where NIR represents the spectral reflectance in near infrared band while R represents red band.

Positive NDVIs represent good vegetation cover, while near zero and negative values indicate degraded vegetation surfaces as well as water, built areas and bare land. Supervised classification was then used to classify the NDVI images into different land use categories giving a total of seven land use classes. This was achieved by making use of seven reflective bands of the images using the maximum likelihood algorithm. The classification maps were further subjected to change detection analysis through cross tabulation; which helped in describing and quantifying differences between images of the same scene at different times. The resultant change maps were then used to calculate the area of the land cover classes identified for each year, followed by an estimation of the change in area occupied by each land cover class over the respective years.

The NDVI images were also used to do cover change detection through the NDVI differencing technique based on subtracting between two NDVI images of the same scene; but of different time periods pixel by pixel to create the difference image. The mean and standard deviation values of the difference image were then used to create the resultant change detection map through the Idrisi re-class module as follows:

(i) **Value 1:** Described areas with significant negative change (overall decrease in vegetation for all land covers) and lying from -1 standard deviation to the lowest NDVI value of the difference image.

(ii) **Value 2:** Areas with normal variation (no significant change in vegetation) and lying from -1 to +1 standard deviation.

(iii) **Value 3:** Described areas with significant positive change (overall increase in vegetation for all land covers) and lying from +1 standard deviation to the highest NDVI value of the difference image.

The relationship between precipitation and NDVI data at different time scales was further investigated. This was done by overlaying points of the rainfall stations on NDVI images for each of the respective years. Five kilometer buffer zones were then created around each station; before digitizing the points on the respective NDVI images, extracting the NDVI values within the buffer and exporting them to Excel. Correlations between NDVI values and annual rainfall trends of the relevant years were then determined in excel to get the relationship between cover change and rainfall for the respective years.

3.4 Results

3.4.1 Annual rainfall variability for the different stations

Time series and trend analysis plots of rainfall variability for Amakhala reserve are shown in Figures 3.2 and 3.7. Both showed low variability in annual rainfall as there was no significant correlation between year and annual rainfall at this station (r = +0.1; Appendix A). The Z scores for Amakhala were so highly variable that it was difficult to discern a clear pattern from them. Similarly, the time series plots for Port Elizabeth (PE) and Port Alfred (PA) did not show any significant trend in annual rainfall variability (Figures 3.2 and 3.7). Most of the 1970s recorded above normal rainfall (positive Z scores), while 1980s and 1990s were characterized by below normal rainfall. Overall correlation analysis did not show a significant trend in annual rainfall (r = -0.25 for PE and -0.22 for PA; Appendix B and C respectively).

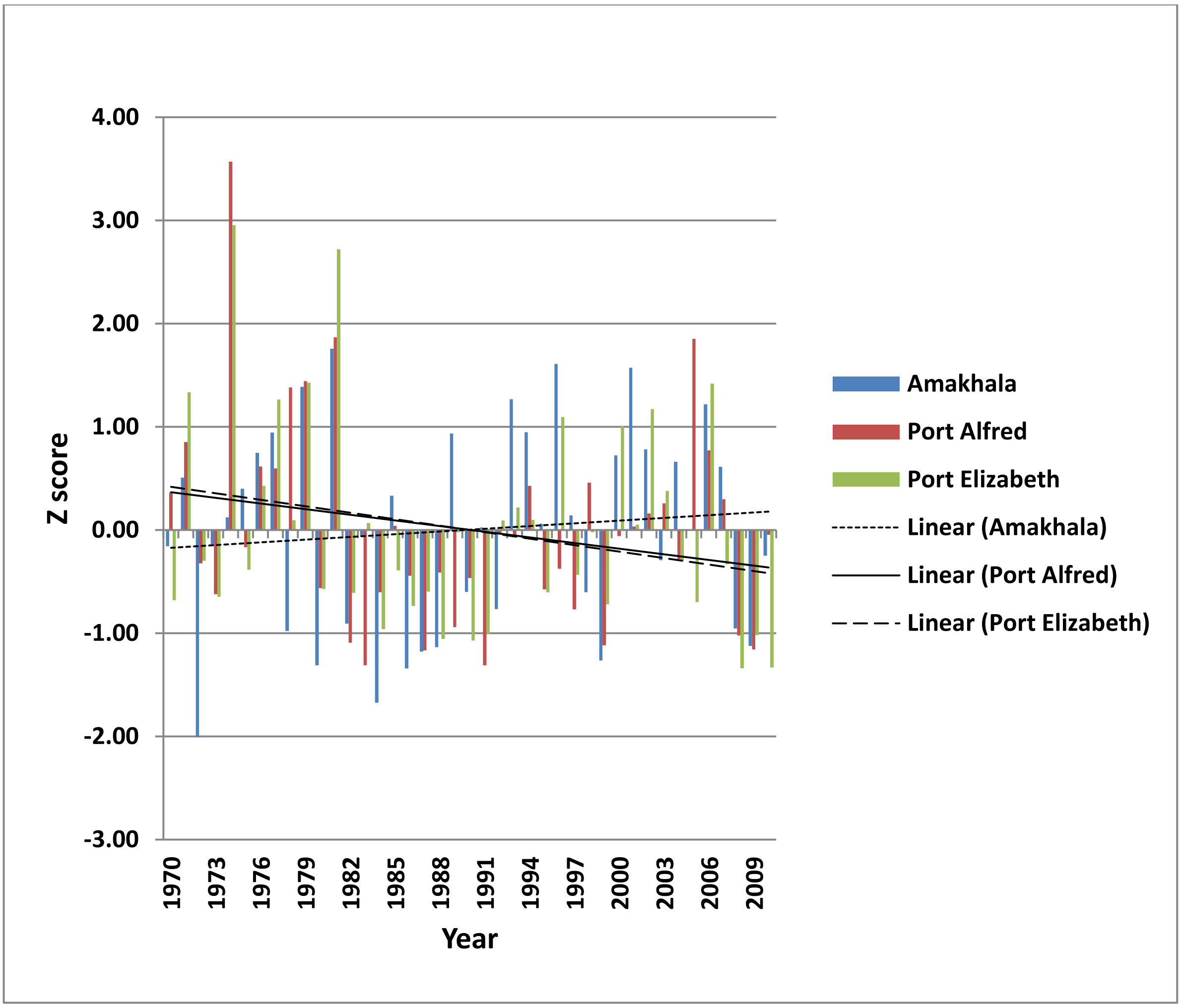


Figure 3.2: Rainfall variability at Amakhala, Port Alfred and Port Elizabeth for 1970-2010

For the studied period of 1970-2001, both Bathurst (Appendix D) and Uitenhage (Appendix E) showed a fairly significant decreasing trend in annual rainfall series ($r = -0.32$ at $p < 0.05$; at both stations). The highest rainfall at either station was received in 1974 (Z score +2.08 for Uitenhage and +2.24 at Bathurst), while the lowest was recorded in 1972 at Uitenhage (Z score = -1.59) and in 1982 at Bathurst (Z score = -1.68). As with the previous two stations; the 1980s and 1990s received below normal rainfall while the 1970s mostly had high rainfall at both stations (Figures 3.3 and 3.7).

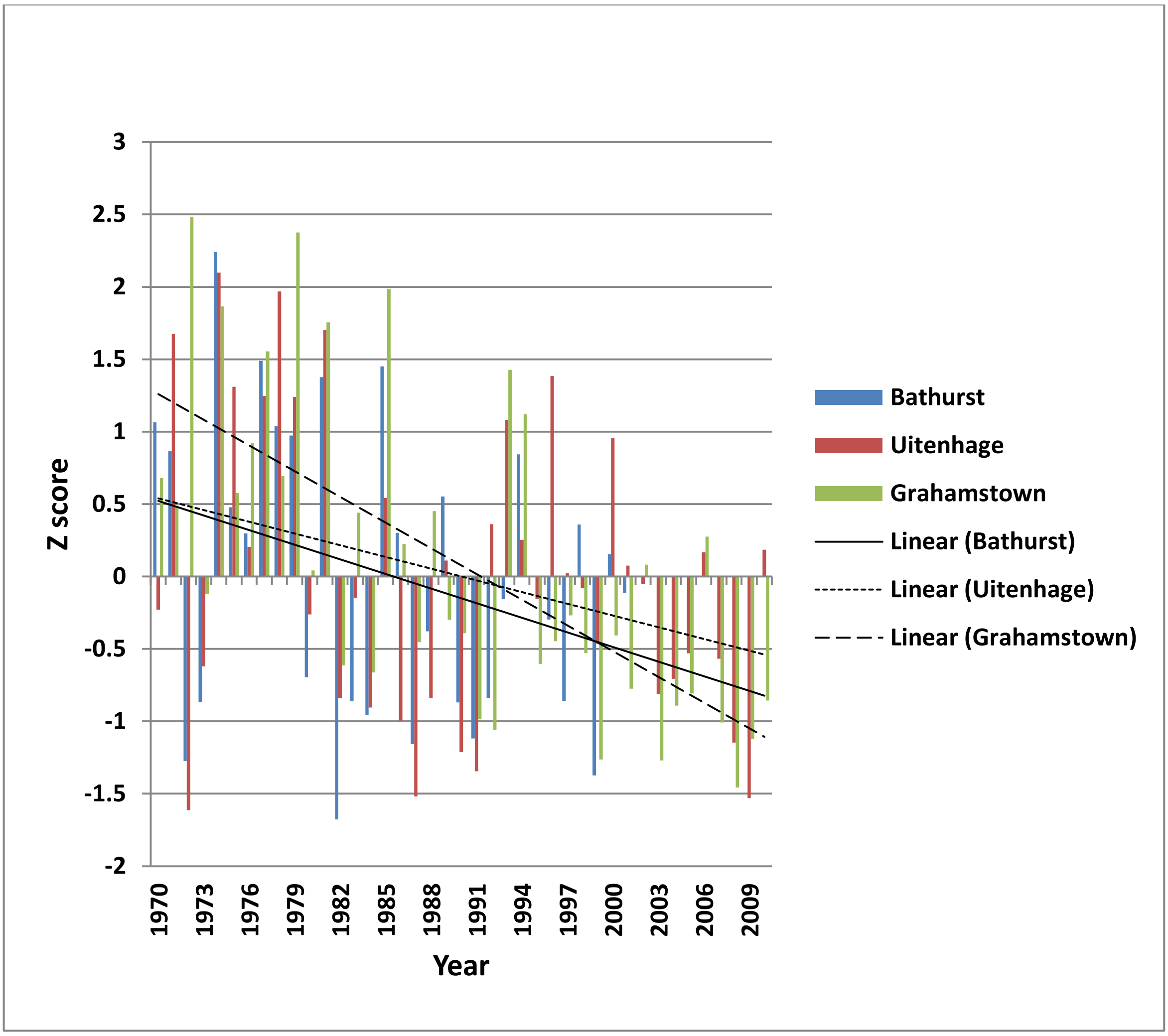


Figure 3.3: Rainfall variability at Bathurst, Uitenhage and Grahamstown for 1970 – 2010

Grahamstown experienced a significant decreasing trend in annual rainfall from the 1970s ($r = -0.59$; $p < 0.05$, Appendix F). The highest rainfall was recorded in 1972 (Z score = +2.48). The period 1989 to 2010 was characterized by below average rainfall with very few years (1993-94, 2002 and 2006) having positive Z scores during this time (Figures 3.3 and 3.7). This was the period mostly responsible for the declining trend. The lowest rainfall at this station was received in 2008 (Z score = -1.46).

3.4.2 Extreme rainfall events for the different stations

As already mentioned, extreme rainfall events were only examined for Grahamstown, Port Alfred, Uitenhage and Bathurst stations (Appendices C – F). A look at the daily rainfall index (DI) showed that it was decreasing over the years (r values recorded at $p < 0.05$ were Grahamstown (-0.57, Appendix F), Port Alfred (-0.51, Appendix C) and Uitenhage (-0.72,

Appendix E). Bathurst however did not show a significant decrease in DI for the years recorded (r = -0.19, Appendix D). The highest rainfall on record has also decreased at Grahamstown over time (r = -0.47) but not for the other 3 stations. Grahamstown, Port Alfred and Uitenhage stations showed an increase in the 0.1-9.9 mm daily rainfall class with time (r = +0.36, +0.58 and +0.36, respectively); thereby showing a negative correlation of this class with DI, (Grahamstown r = -0.74, Port Alfred = -0.71; Bathurst = -0.46 and Uitenhage = -0.77). This rainfall class also related negatively with the frequency and length of the dry spells for most of the stations. The 10-19.9 mm class decrease over the years was significant for Grahamstown and Uitenhage (r = -0.51 in both instances); while the > 19.mm category decreased significantly for Grahamstown (r = -0.39); Port Alfred (r = -0.43) and Uitenhage stations (r = -0.43). The 10-19 mm as well as the > 19.9 mm classes closely followed the annual rainfall and DI patterns at most stations. The frequency of dry days showed a decreasing trend over time at Port Alfred and Uitenhage stations; while length of dry spells increased only at Bathurst station (r = +0.41).

3.4.3 Seasonal annual rainfall patterns for the stations

Generally, the areas studied receive rainfall throughout the year; with winter rain falling from June to August (Figure 3.4). Spring rain falls from September to November, summer rainfall is from December to February and autumn rain falls in March to May. This trend is clearly illustrated by the smoothened two year running mean plot in Figure 3.4. The highest rainfall was received in November in most years while June recorded the lowest rainfall.

Seasonal annual rainfall variations showed a similar decreasing trend from the 1970s through the 1980s and 1990s as with annual rainfall, Figure 3.5, (hence they had high positive correlations with annual rainfall for all seasons). Autumn and winter annual rainfalls were especially responsible for the sharp declines in the 1980s and 1990s; while spring annual rainfall showed little variation over the years.

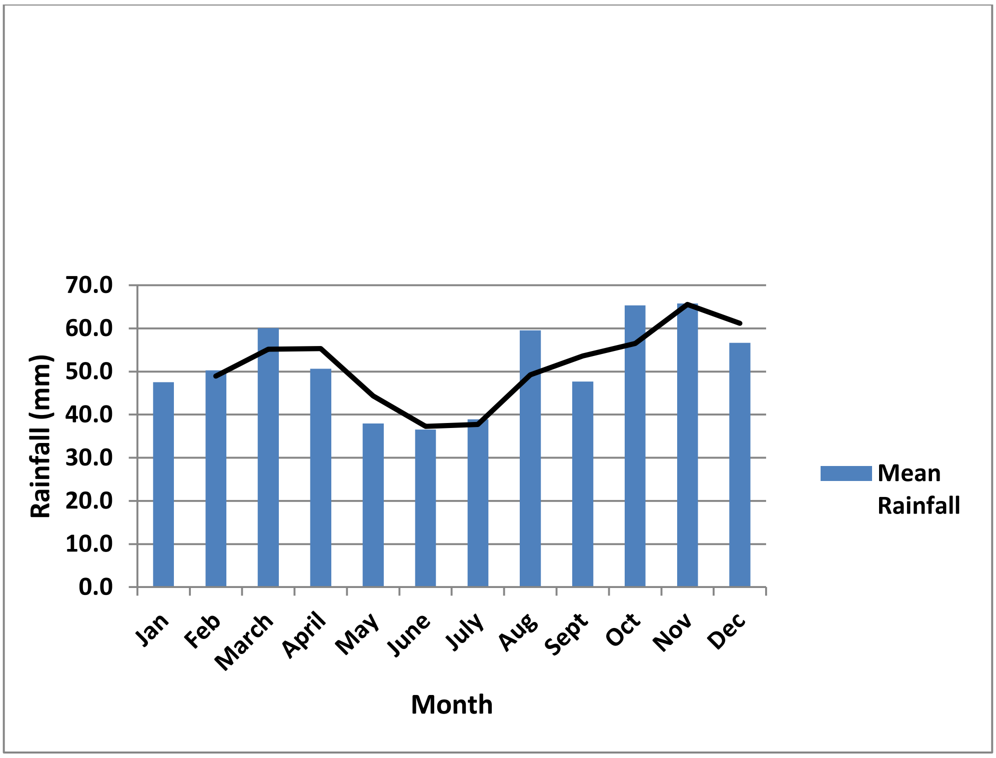


Figure 3.4: Mean monthly rainfall for all stations for 1970 - 2010

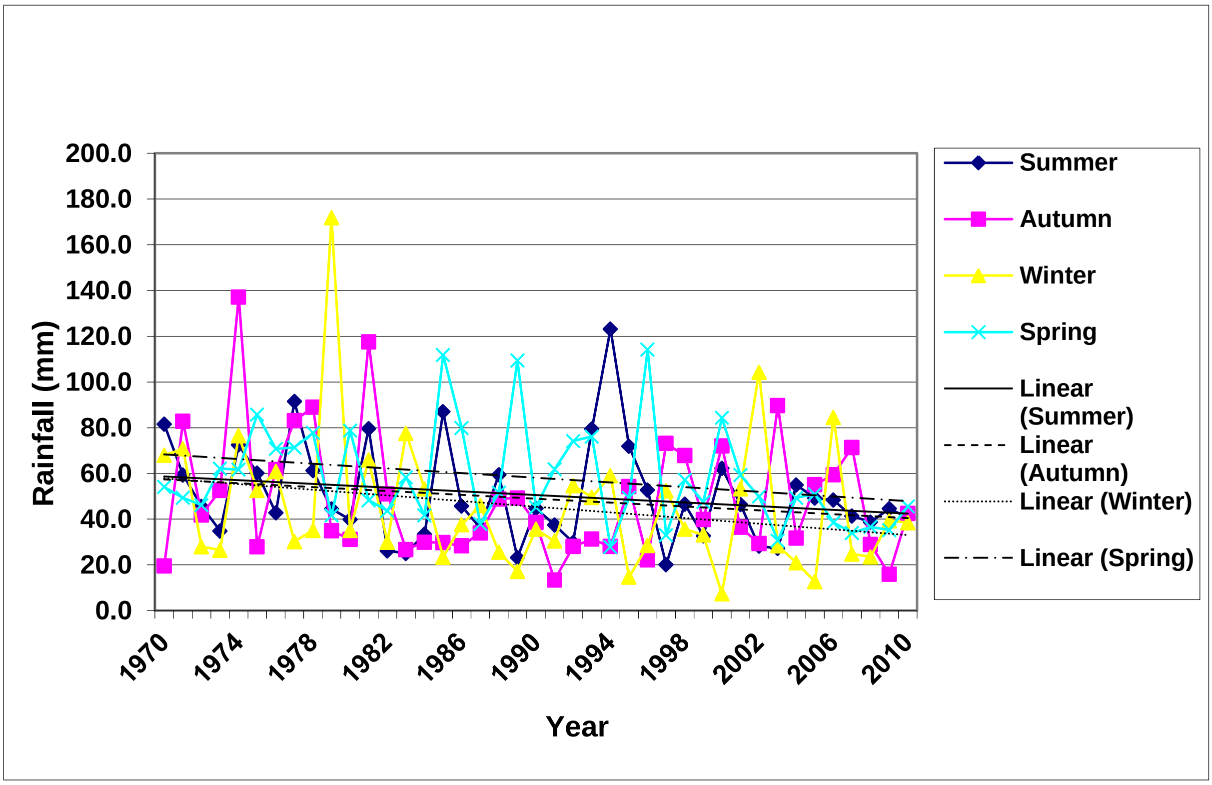


Figure 3.5: Mean seasonal rainfall variability for all stations for 1970 - 2010

3.4.4 Comparisons between stations

The combined mean plot of annual rainfall for all stations (Figure 3.6) showed a significant long term decreasing trend in annual Rainfall (r = -0.41 and $R^2 = 0.17$ from Regression analysis at $p < 0.05$, Table 3.1). Regression analysis of the individual stations and the combined mean annual rainfall plots also followed the same trend as that obtained from correlation analysis. Thus Grahamstown experienced the most significant decrease in rainfall over the years ($R^2 = 0.35$) followed by Uitenhage and Bathurst ($R^2 = 0.1$), while Port Elizabeth, Port Alfred and Amhakala had no significant trends (Table 3.1). Rainfall comparisons showed Amakhala station to have received consistently lower rainfall throughout the years (Figure 3.7 and Table 3.1). The variance in rainfall is highest for Grahamstown and lowest at Amakhala as noted from the standard deviations (Table 3.1). High correlations were observed between all stations studied; with consistent similarities in rainfall patterns over time i.e.; the 1970s receiving high rainfall amounts followed by a decreasing trend thereafter (Appendix G).

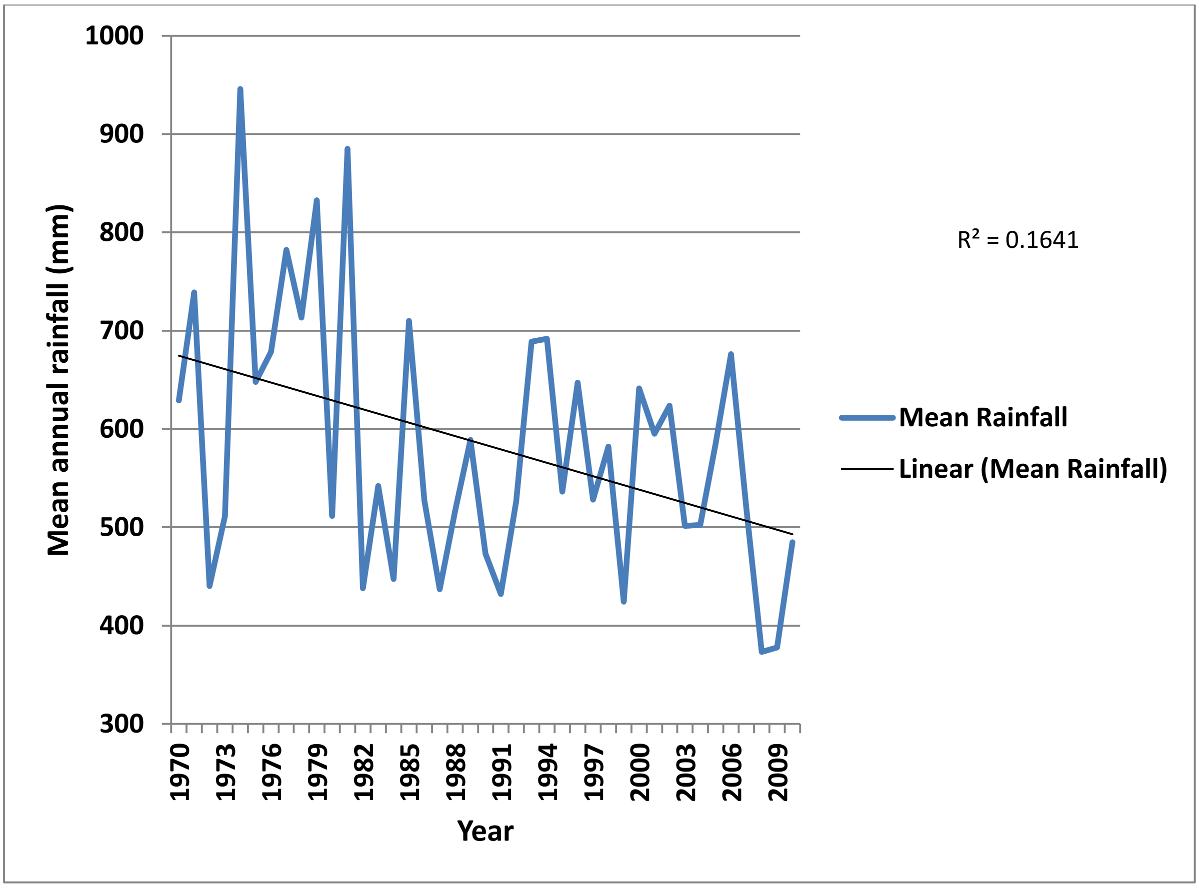


Figure 3.6: Mean annual rainfall variability for all stations for 1970 - 2010

Table 3.1: Annual rainfall (mm) summaries for the stations

	Amakhala	Port Elizabeth	Grahamstown	Port Alfred	Uitenhage	Bathurst	Combined Stations
Mean	443.1	605.3	649.4	645.7	471.2	731.8	602.8
Minimum	216.3	401	348.8	421.7	244.5	450.0	424.4
Maximum	641.8	1055.8	1139.6	1254.7	765.9	1107.4	945.8
Standard Deviation	111.7	154.3	206.4	170.7	142.2	168.0	137.4
No. of Years studied	41	41	41	41	41	32	41
Correlation with Years ($p < 0.05$)	+0.1 ^{ns}	-0.25 ^{ns}	-0.59**	-0.22 ^{ns}	-0.32*	-0.32*	-0.41*
Regression (R^2 value)	0.01 ^{ns}	0.06 ^{ns}	0.35**	0.05^{ns}	0.1*	0.1*	0.17*

***Very significant, **Significant, *low significance, ^{ns} not significant

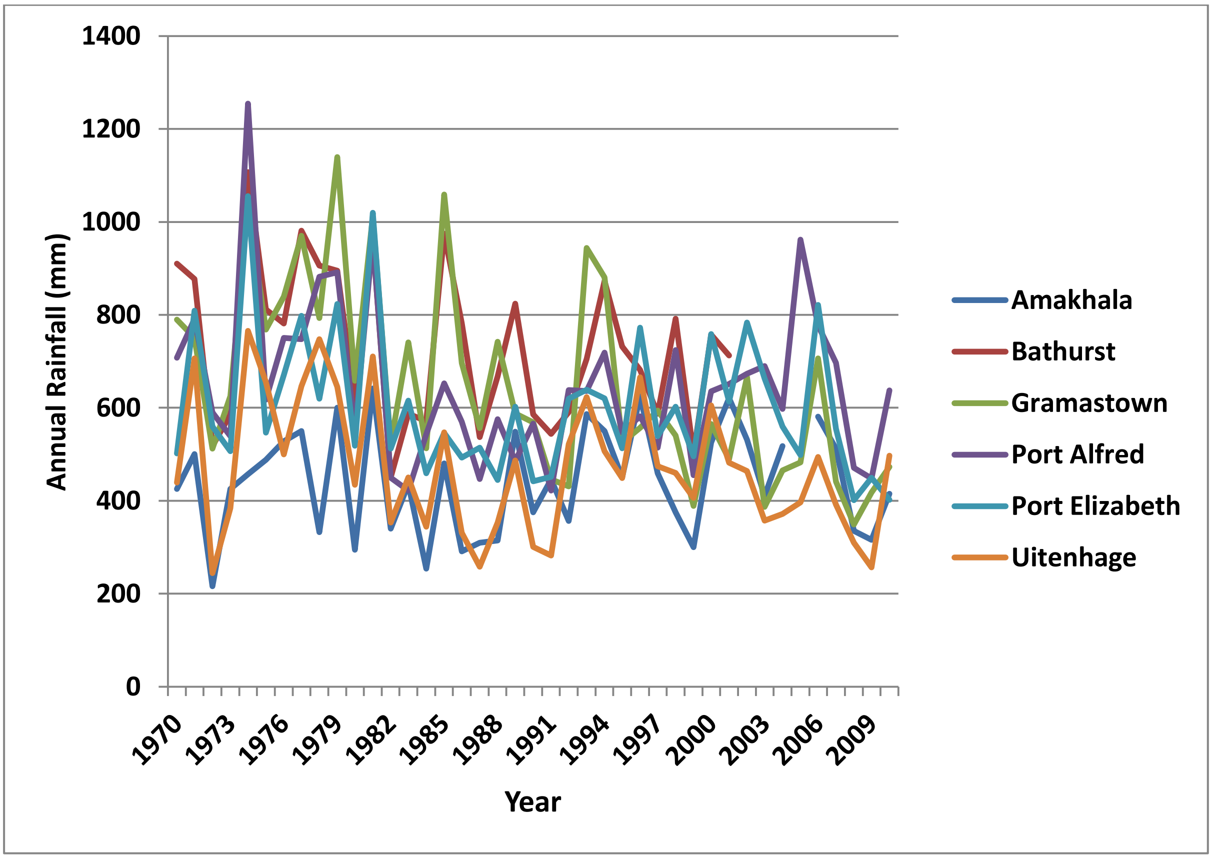


Figure 3.7: Time series plot of annual rainfall for different stations for 1970-2010

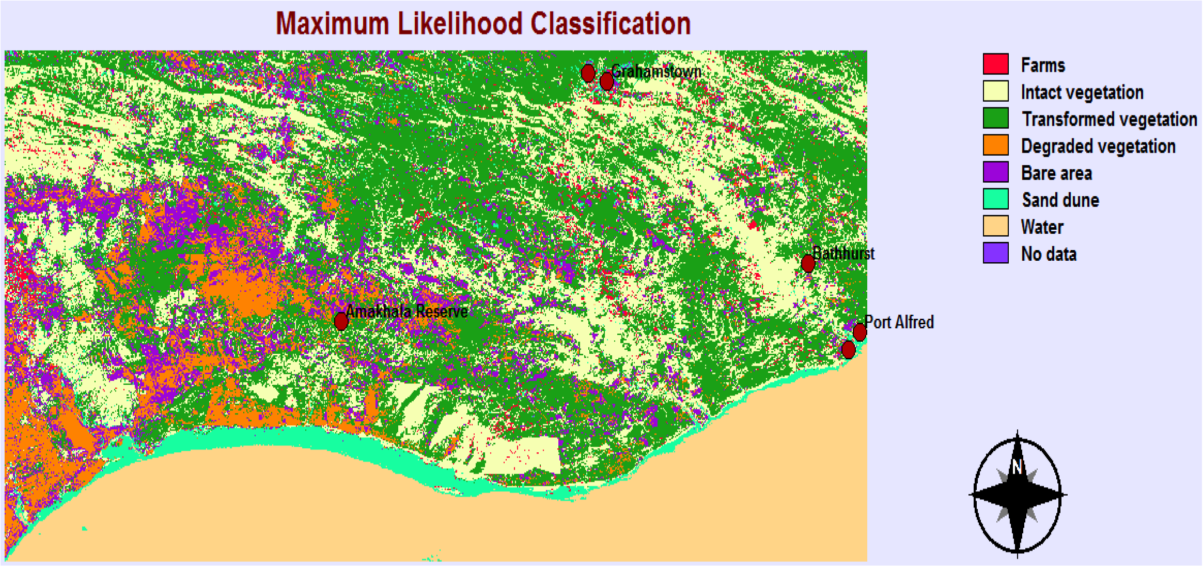
3.4.5 Variation in LUC change at the study sites

The first stage involved a general LUC classification of the area occupied by the six rainfall stations. Table 3.2 below gives a description of the seven LUC classes observed from satellite images. In some images, an eighth class was included, which represented the area without any data available that usually lied just outside the image boundary.

Table 3.2: Land cover class descriptions

Land cover class	**Description**
Farms	Mostly cultivated commercial farms with lash vegetation
Intact vegetation	Dense vegetation
Transformed vegetation	Moderately degraded vegetation
Degraded vegetation	Degraded areas with very sparse vegetation
Bare areas	Areas without any vegetation cover + uncultivated land
Sand dune	Area at the edge of the sea covered in sand
Water	Area covered by water bodies i.e. the sea, rivers, dams

The LUC classification results for the different years are shown in Figures 3.8 to 3.10 and summarized in Table 3.3. Between 1989 and 1999, there was a substantial decrease in the area occupied by intact vegetation by about 137 860 ha (i.e. an overall decrease of 17.4 % in comparison with the other LUC classes), while the area occupied by farms and sand dunes decreased slightly by 17 095 ha (1.9 %) and 6 148 ha (1.1 %), respectively. Increases of 133 794 ha (16.3 %) and 51 316 ha (5 %) were recorded in the area occupied by degraded and transformed vegetation, respectively; while the bare areas (2 129 ha) and those occupied by water (3 615 ha) remained more or less unchanged (~ 0 %) between 1989 and 1999 (Figure 3.8, Table 3.3).



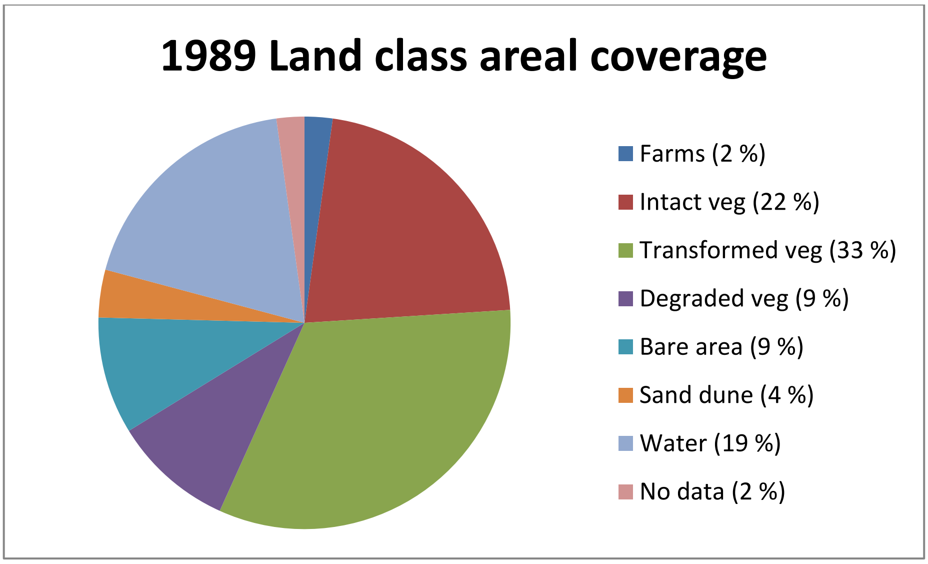


Figure 3.8: Land use / cover map for 1989 and the areal coverage for its LUC classes

When this analysis was done over a longer 20 year period (i.e. 1989 to 2009), the changes previously recorded for area occupied by farms (970 ha, 1.8 %), sand dunes (575 ha, 1.2 %) and water (3 065 ha, 0.1 %) during the 1989-1999 period remained more or less the same (Figure 3.9, Table 3.3). What differed were the changes in areal coverage of degraded and transformed vegetation which increased by smaller margins of 68 743 ha (8.5 %) and 23 965 (1.7 %), respectively between 1989 and 2009 compared with the 1989-1999 increases. The bare areas however decreased by 41 655 ha (5.1 %) between 1989 and 2009.

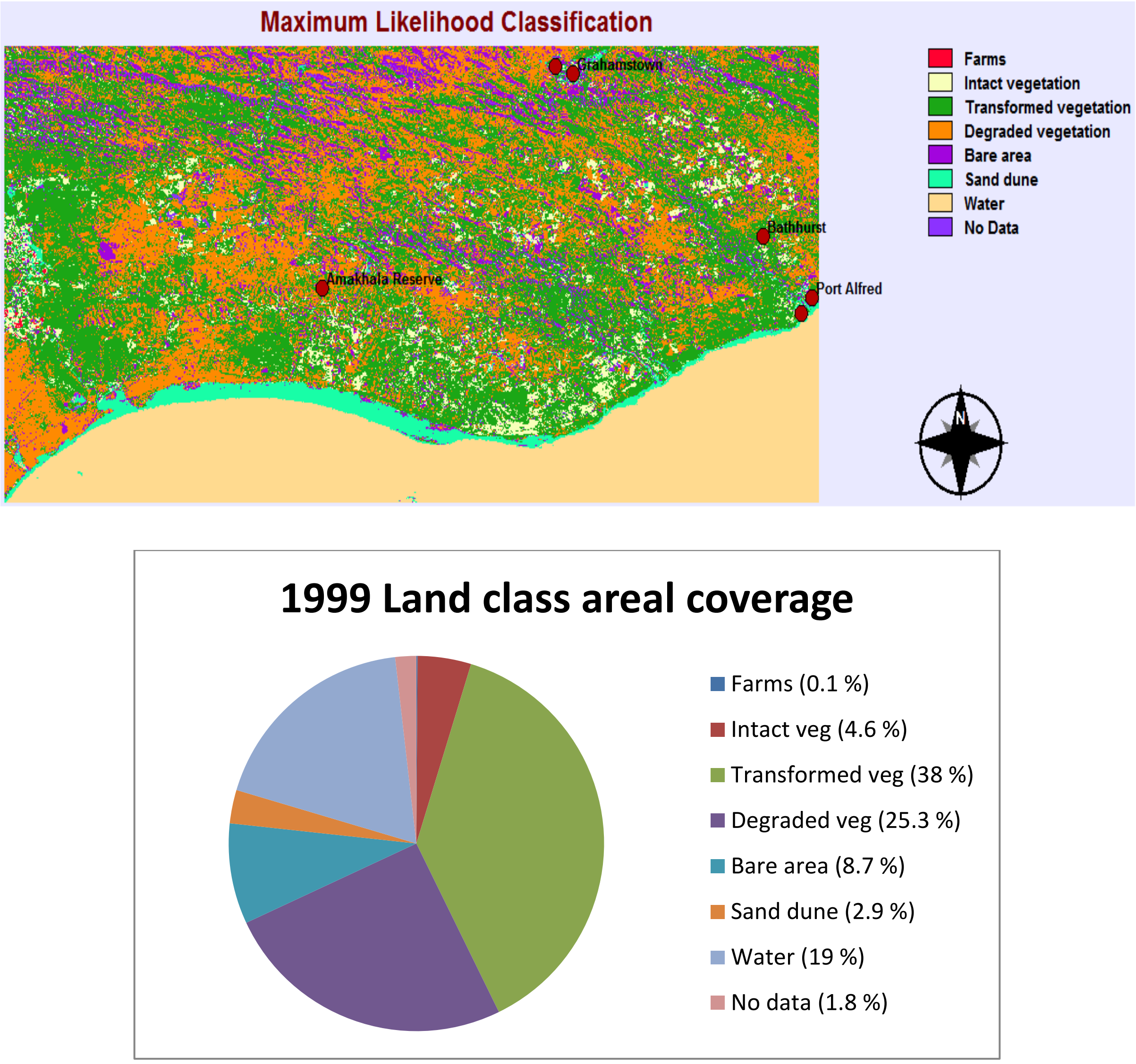


Figure 3.9: Land use / cover map for 1999 and the areal coverage for its LUC classes

A comparison of the LUC classes between 1999 and 2009 showed a substantial increase in the area occupied by intact vegetation by about 144 094 ha (17.4 %); while the degraded, bare and transformed vegetation area decreased by 68 743 ha (7.8%), 39 526 ha (4.8 %) and 27 352 ha (3.3 %), respectively (Table 3.3). The area occupied by water and sand dunes also remained unchanged (~ 0 %) between 1999 and 2009.

A closer look at change detection between different time periods showed that some gains that were made in one period, were compensated for by losses made in a subsequent period and vice–versa resulting in major shifts in overall LUC changes over the twenty year study period (Table 3.3). A good example of this is the 17.4 % (137 860 ha) loss in intact vegetation cover experienced between 1989 and 1999; which was accompanied by a 17.4 % (144 094 ha) gain in this same LUC class between 1999 and 2009, resulting in a net change of 0% between 1989 and 2009. Similarly transformed and degraded vegetation that resulted in areal gains of 5 % (51 316 ha) and 16.3 % (133 794 ha), respectively in the 1989-1999 sub-period, lost 3.3 (27 352 ha) and 7.8 % (65 051 ha) of this vegetation cover in 1999-2009 sub-period, resulting in overall gains of 1.7 (23 694ha) and 8.5 % (68 743), respectively between 1989 and 2009.

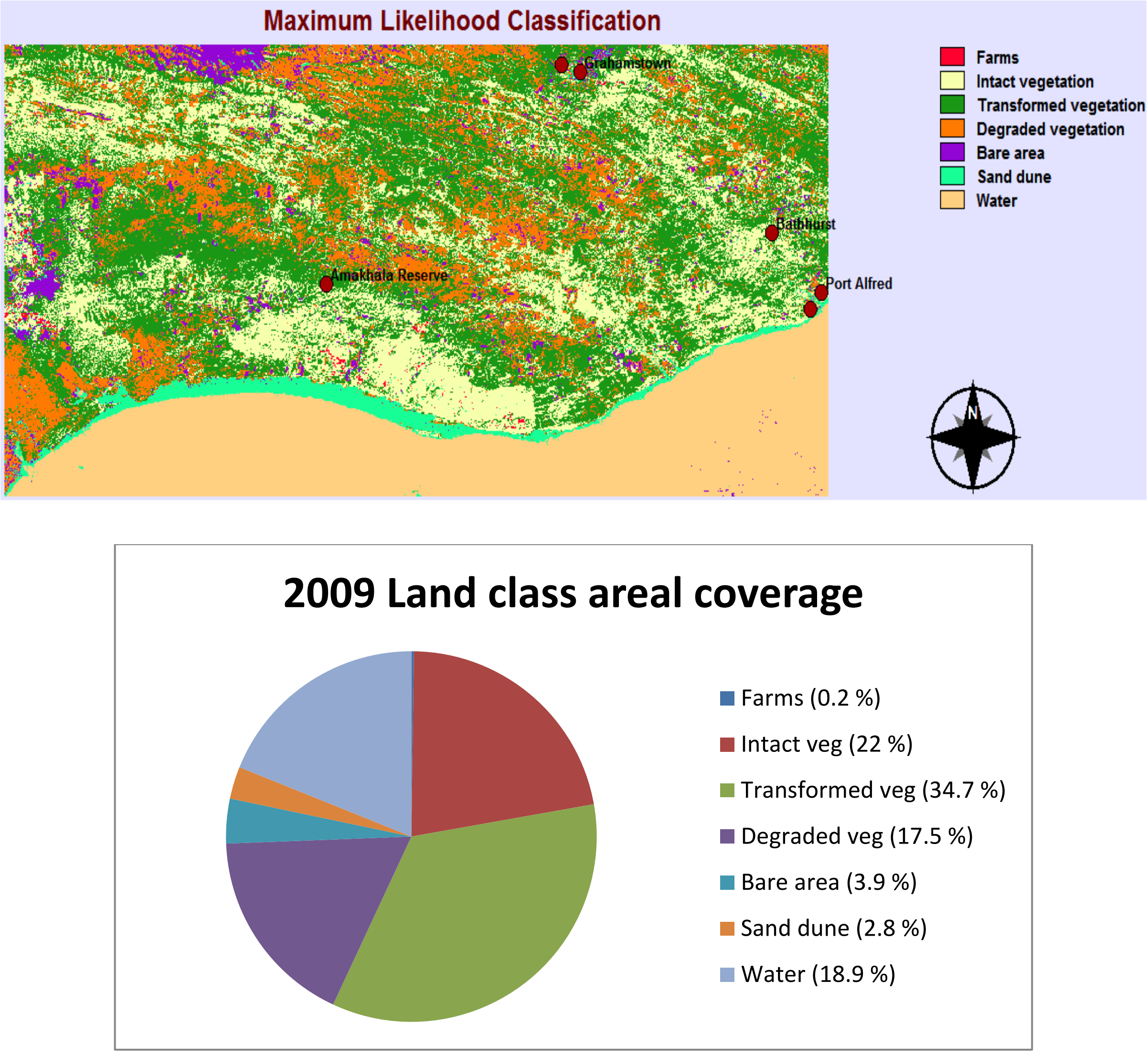


Figure 3.10: Land use / cover map for 2009 and the areal coverage for its LUC classes

Farmland that decreased by 1.9 % (17 095 ha) in 1989-1999, remained more or less unchanged between the 1999-2009 sub-period (970 ha, 0.1 %) resulting in a net loss of 1.8 % (16 125 ha) from 1989-2009 (Figure 3.10, Table 3.3). On the other hand, bare areas that decreased slightly (2129 ha, 0.3 %) between 1989-1999, decreased by 4.8 % (39 526 ha) in the 1999-2009 sub-period, resulting in an overall decrease of 5.1 % (41 655 ha) from 1989 to

2009; while the area occupied by sand dunes that decreased by 1.1 % (6 148 ha) in 1989-1999 but remained unchanged (576 ha, 0.1 %) in 1999-2009 had an overall decrease of 1.2 % (6 723 ha) between 1989 to 2009. The area occupied by water did not vary much over the 20 year study period.

Table 3.3. Summary of LUC classification area for 1989, 1999 and 2009

LUC class	**1989**		**1999**		**2009**		**1989-1999**		**1989-2009**		**1999-2009**	
	ha	**%**	**ha**	**%**	**ha**	**%**	**ha**	**Relative Change (%)**	**ha**	**Relative Change (%)**	**Ha**	**Relative Change (%)**
Farms	18000	2	905	0.1	1875	0.2	-17095	-1.9	-16125	-1.8	+970	+0.1
Intact vegetation	176000	22	38140	4.6	182234	22	-137860	-17.4	+6234	0	+144094	+17.4
Transformed vegetation	264000	33	315316	38	287965	34.7	+51316	+5	+23965	+1.7	-27352	-3.3
Degraded vegetation	76000	9	209794	25.3	144743	17.5	+133794	+16.3	+68743	+8.5	-65051	-7.8
Bare areas	74000	9	71870	8.7	32345	3.9	-2129	-0.3	-41655	-5.1	-39526	-4.8
Sand dune	30000	4	23852	2.9	23277	2.8	-6148	-1.1	-6723	-1.2	-575	-0.1
Water	150000	19	153615	18.6	156680	18.9	+3615	-0.4	+6680	-0.1	+3065	+0.3
No Data	18000	2	15017	1.8	-	-	-2983	-0.2	-	-	-	-

3.4.6 Cross-tabulation of the LUC classes

The cross-tabulation analysis of the LUC classes gave a more detailed examination of the shifts in areal coverage of the different land cover classes. The 1989-1999 cross tabulation for example showed that some of the land that used to be occupied by farms and intact vegetation in 1989 was taken up by bare areas, degraded and transformed vegetation (Kappa = 0.25 and Cramer’s V = 0.46, Table 3.4). Part of the area for transformed vegetation was re-assigned to intact, bare land and degraded vegetation in 1999, while some degraded areas were taken up by transformed, intact and degraded vegetation. The areas that used to be bare in 1989 were partly taken up by transformed, intact and degraded land, while some of the sand dune area was re-assigned to bare land, degraded and transformed vegetation. Very little change was observed on the area occupied by water. It can be observed from this analysis that the degraded vegetation class invaded into most of the LUC classes which could explain the substantial net increase (16.3 %) in degraded land between 1989 and 1999. Transformed land also invaded other LUC classes but to a lesser extent; resulting in a net increase of 5 %. The area previously occupied by intact vegetation and farms was converted to transformed, bare and degraded land leading to substantial losses of these two classes by 17.1 and 1.9 %, respectively between 1989 and 1999.

Table 3.4. Cross-tabulation of 1989 classification (columns) against 1999 classification (rows)

```
Farm    Intact   Transfmd  Degraded  Bare    Sand dune  Water    No data    Total
       -------------------------------------------------------------------------------------
Farm       1403        344     1130      2728      3878       301        4        169     9957
Intact    35749      62229   141795     57730     95056     12324       51      14756   419690
Transfmd 121603    1231766  1249248    292763    410483     58344      601     103388  3468196
Degraded  22983     328494  1254318    422648    228447     25524      106      24761  2307281
Bare      11076     284985   298831     68322     62091     36957     1953      25714   789929
Sand dune  4628       9994    11869     14842     20625    188585     8512       3798   262853
Water        21         16       34       236        82     10945  1683464          0  1694798
No data    2710      70301    28331      5616     16581      8063     6594      26782   164978
       ---------------------------------------------------------------------------------------
Total  200173    1988129  2985556    864885    837243    341043  1701285     199368  9117682

Chi Square = 13543939.00000
              df =    49
         P-Level =    0.0000
      Cramer's V =    0.4607
     Overall Kappa   0.2542
```

Between the longer 1989 to 2009 period, Table 3.3 shows that there was reduction in bare areas (5.1 %, 41 655ha) and farm land (1.8 %, 16 125ha). The cross tabulation shows that most of the farm land was taken up by transformed, bare and degraded LUC classes; while the bare areas were mostly occupied by transformed, degraded and intact vegetation (Kappa = 0.46 and Cramer's V = 0.52, Table 3.5). Gains were made in degraded (8.5 %, 68 743ha) and transformed (1.7 %, 23 965ha) LUC classes although they were not as high as those made during the 1989-1999 sub-period. The degraded LUC class increased since it took up some of the farms, transformed, bare and some sand dune area; while the transformed vegetation was re-assigned to some farms and intact vegetation.

Table 3.6 shows that net losses were observed in the area occupied by degraded (7.8 %, 65 051ha), bare (4.8 %, 39 526ha) and transformed (3.3 %, 27 352ha) LUC classes between 1999 and 2009 (Kappa = 0.32 and Cramer's V = 0.51). Part of the area occupied by transformed vegetation was taken up by degraded vegetation, bare area and intact vegetation; while degraded land was partly taken up by transformed, bare areas and intact vegetation and the bare areas made way to transformed, degraded and intact vegetation. This caused the intact vegetation to experience major increases of up to 17.4 % (144 094ha) in 2009 in comparison to 1999. On the whole, the intact vegetation was reassigned to most of the LUC classes such as farms, transformed, degraded and bare areas.

Table 3.5: Cross-tabulation of 1989 classification (columns) against 2009 classification (rows)

	Farm	Intact	Transfmd	Degraded	Bare	Sand dune	Water	No data	Total
Farm	**2979**	2277	5814	3891	4507	607	0	556	**20631**
Intact	69969	**1236128**	479216	48978	75587	30089	3505	59384	**2002856**
Tranfmd	84983	594811	**1704948**	314011	334891	55109	379	75199	**3164331**
Degrded	28211	93256	681082	**406824**	304483	37250	423	39334	**1590863**
Bare	9995	27317	101856	77169	**100827**	17557	1643	19088	**355452**
Sandune	3985	14718	11807	13321	16519	**184336**	8353	3294	**256333**
Water	51	19622	833	691	429	16095	**1686982**	2513	**1727216**
Total	**200173**	**1988129**	**2985556**	**864885**	**837243**	**341043**	**1701285**	**199368**	**9117682**

Chi Square = 15085837.00000 df = 42 P-Level = 0.0000
Cramer's V = 0.5251 Overall Kappa 0.4677

Table 3.6: Cross-tabulation of 1999 Classification (columns) against 2009 classification (rows)

	Farm	Intact	Transfmd	Degraded	Bare	Sand dune	Water	No data	Total
Farm	**2068**	9655	5696	834	1303	615	0	460	**20631**
Intact	3621	**139596**	1154101	354790	267845	16964	41	65898	**2002856**
Transfmd	1433	140820	**1546594**	1145133	271493	15139	76	43643	**3164331**
Degraded	711	81356	584218	**710361**	176510	15286	82	22339	**1590863**
Bare	1596	39160	153362	84975	**43105**	21740	1612	9902	**355452**
Sanddune	527	8918	22318	10361	19638	**183362**	7293	3916	**256333**
Water	1	185	1907	827	10035	9747	**1685694**	18820	1727216
Total	**9957**	**419690**	**3468196**	**2307281**	**789929**	**262853**	**1694798**	**164978**	**9117682**

Chi Square = 14078785.00000
df = 42
P-Level = 0.0000
Cramer's V = 0.5073
Overall Kappa 0.3191

3.4.7 Variations in NDVI over the years

Table 3.7 below gives a summary of the NDVI values for the images of the different years and their overlays obtained through the NDVI differencing technique. It can be observed that the 1989 image had a wider NDVI range of -1 to 0.841 showing a wider diversity of both bare and the densely vegetated areas in this year; while the 1999 image had the least NDVI range of -0.5 to 0.677 and the lowest mean NDVI of 0.098; indicating the least amount of vegetation cover in this year. The 1989-2009 overlay showed the widest NDVI range of -1.676 to 0.945 and the least mean NDVI value of -0.039, indicating a greater diversity in vegetation cover in this period. The change detection maps of the overlays showed the area around and to the west of Amakhala reserve to have experienced the greatest change in vegetation cover with more losses in cover experienced in the intact, transformed and farm land within this region over the years. Areas around Grahamstown had the greatest increase in transformed vegetation cover over the years, while Bathurst and Port Alfred had variable vegetation cover changes over time.

Table 3.7: Descriptive statistics for NDVI values

	Minimum	Maximum	Range	Mean	StDev
1989 NDVI	-1	0.841	1.841	0.172	0.247
1999 NDVI	-0.5	0.677	1.177	0.098	0.198
2009 NDVI	-0.571	0.783	1.357	0.211	0.229
1989-1999 Overlay	-1.536	0.93	2.466	0.073	0.133
1989-2009 Overlay	-1.676	0.945	2.621	-0.039	0.132
1999-2009 Overlay	-0.945	0.983	1.928	0.113	0.101

The actual areal coverage of the resultant image obtained from the NDVI change detection analysis is shown in Figure 3.11. As can be observed, most of the area covered by the images remained relatively unchanged in all three sub-periods; with the 1989-2009 period having the largest area of vegetation remaining unchanged (635 000 hectares, 79 %), while experiencing the least significant losses (86 000 hectares, 11 %) and gains (84 000 hectares, 10 %) in vegetation. A trend which is similar to the LUC class change analysis obtained before. The 1999-2009 period however experienced the most significant losses (106 746 hectares, 12.9 %) and gains (104 277 hectares, 12.6 %) in vegetation causing it to have the least amount of vegetation cover (617 487 hectares, 74.5 %) that remained unchanged. The 1989-1999 sub-period had 11 % (92 000 hectares) significant loss, a 12 % (94 000 hectares) gain in vegetation and 77 % (619 000 hectares) of its area having vegetation that remained unchanged.

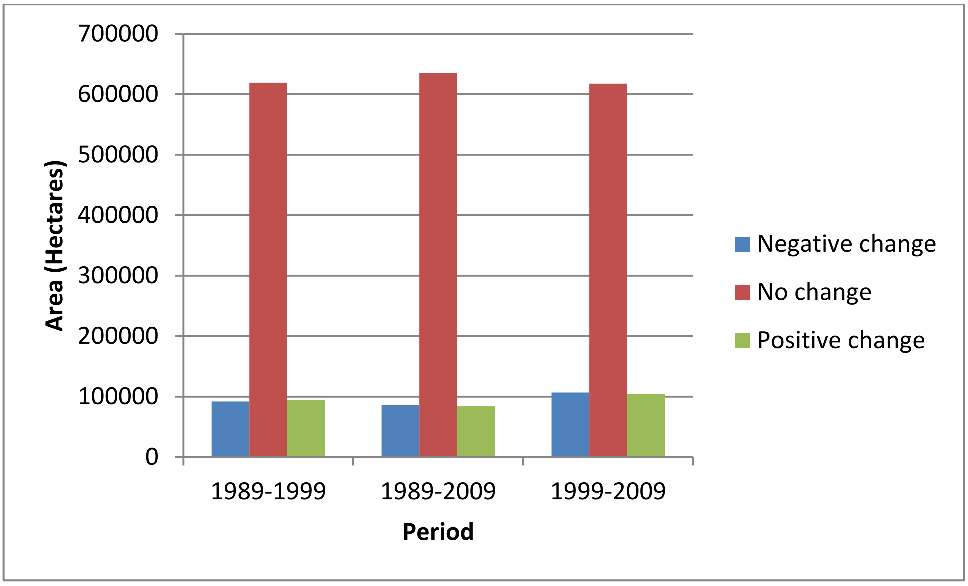


Figure 3.11: Areal coverage of the NDVI change detection maps for 3 sub-periods

3.4.8 Relating NDVI with rainfall variability over the years

A correlation analysis between annual rainfall and NDVI values within 5 km buffer zones for each stations and respective year showed the 1989 NDVI maximum, mean and sum to increase with increasing rainfall amount (r = +0.4, +0.33 and +0.65 respectively @ $p < 0.05$ from correlation analysis; Table 3.8). In 1999 however, the minimum NDVI decreased with increasing rainfall (r = -0.71) while the mean NDVI and its range increased with increasing rainfall (r = +0.22 and +0.54 respectively) and the other NDVI values had no significant relationship with rainfall. In the case of 2009, the minimum, maximum, mean and NDVI sum all decreased with increasing rainfall (r = -0.31, -0.4, -0.33 and -0.73 respectively). As discussed before, there was a general decline in rainfall in 1980s and 1990s at most stations with Grahamstown showing a particularly heavy decline between 1989 and 2009 (Figures 3.2 and 3.3). The correlation was especially strong in 1999 which was characterized by low rainfall at most stations and had correspondingly the lowest NDVI mean and sum (at 5 out of 6 stations); and lowest maximum NDVI values at all stations. The most significant NDVI – rainfall trends were observed at the two Grahamstown stations that had the lowest NDVI values in 1999 when rainfall was lowest, the highest NDVIs in 1989 when rainfall was highest and moderate NDVI values in 2009 when rainfall amounts were moderate.

Table 3.8: Correlation (r value) between NDVI and annual rainfall for 6 stations ($p < 0.05$)

NDVI	**1989**	**1999**	**2009**
Minimum	-0.03	-0.71***	-0.31*
Maximum	+0.40*	-0.14	-0.40
Mean	+0.33*	+0.22	-0.33*
Range	+0.13	+0.54**	+0.15
Sum	+0.65**	+0.12	-0.73***

***Very significant ($p < 0.001$), **Significant ($p < 0.01$), *low significant ($p < 0.05$)

3.5 Discussion

3.5.1 Trends in rainfall variability

Long term variability in annual rainfall can be seen with a declining trend at Grahamstown Uitenhage and Bathurst stations (Figure 3.3). No clear variability in rainfall was observed for Amakhala Reserve, Port Alfred and Port Elizabeth (Figure 3.2). Five of the stations studied and the pooled rainfall for all stations showed a significant declining trend in the 1980s and 1990s sub-period, with the 1970s being significantly wetter years. A study by Blignaut (2009) on rainfall variability between 1970 and 2006 also showed a trend of South Africa becoming drier between 1997 and 2006 compared to the 1970s in eight of the nine provinces studied. Valimba (2004) identified significant shifts in the number and amounts of annual daily rainfall events (especially in the < 10 mm class) in the 1970s, with a decreasing trend in February-May and an increasing trend in October-January. Nicholson (2000) reported that rainfall variability over east and southern Africa was mostly influenced by the El Niño Southern Oscillation (ENSO) cycle and sea surface temperatures (SSTs) over the Atlantic and Indian Oceans. This ENSO can manifest itself as either El Niño or La Niña associated with warm and cool seas surface temperatures respectively. Nicholson (2000) further explained that the ENSO influence was cyclic in nature and strongest for inter-annual variability with each cycle including periods of above-average and below-average rainfall. Kane (2009) reported two giant El Niño events in 1982/83 and 1997/98; which were responsible for the extensive 1982/83 droughts in the summer rainfall regions of South Africa. The 1997/98 effects however did not result in significant droughts. Washington and Preston (2006) also showed evidence of La Nina episodes that prevailed over the Pacific Ocean, and were characterized by unusually high summer rainfall in January to March of 1974 and 1976 in southern African. Likewise, Indian Ocean SSTs characterized by warm anomalies in the subtropical SW Indian Ocean and cool anomalies in the northern SW Indian Ocean were associated with the two wettest years (i.e. 1974 and 1976) of the twentieth century in Southern Africa (Washington and Preston, 2006). Thus the observed declining trends from the 1970 and the 1980s -1990s sub-periods could possibly be explained by these ENSO episodes.

It can be observed that it is sometimes difficult to discern clear patterns from annual rainfall variability. Williams *et al*. (2011) reported that changing climate variability and climate extremes may have a greater impact on environmentally vulnerable regions than a changing annual mean rainfall. Moreover, climate-related extremes have been the dominant trigger of natural disasters such as droughts, floods and disease epidemics in southern Africa over the past decades (Shongwe *et al*., 2009). When studying extreme weather events, Shongwe *et al*., (2009) observed a delay in the onset and early cessation of the rainy season, as well as a projected decrease in precipitation in the hyper-arid and semi-arid areas of southern Africa. It is therefore important to study daily rainfall variability (instead of concentrating on mean annual rainfall variability alone) since it helps explain extreme weather events. Less attention has been paid to the study of extreme weather events in the past. This was partly due to the inadequacies of extreme weather and daily rainfall data in African climate studies (Christensen *et al*., 2007). In the present study, daily data were only available for Grahamstown, Port Alfred, Uitenhage and Bathurst stations.

A closer look at the daily data showed the daily rainfall index (DI) as well as most of the subclasses from 10-19.9 mm and above to be decreasing over the years for Grahamstown, Port Alfred and Uitenhage stations. It appeared the frequency of wet days had a greater impact on rainfall variability than the frequency of dry days. This was shown by the highly positive correlation between the 10-19 mm and the > 19.9 mm daily sub-classes with annual rainfall and DI patterns. Shongwe *et al*. (2009) reported pronounced increases in heavy precipitation events to be expected where mean total seasonal or annual precipitation increased; or dry extremes to be more severe where mean precipitation decreased. Dollar and Rowntree (1995), also noted both the magnitude and frequency of events to be greater for wetter than drier periods. They observed wet years to be a result of an increase in frequency of daily events rather than an increase in the magnitude or intensity of the wet event. In the present study, both the frequency and size of wet events contributed significantly to annual rainfall as seen from the highly positive correlations between the rainfall classes of 10 mm and above with both DI and annual rainfall. Dry spells only showed an increasing trend over the years at Bathurst station. More studies on extreme weather events that have been done in the southern African region include the devastating floods observed over southern Mozambique or NE South Africa in early 2000 and the intense drought over much of Zambia, Malawi, Zimbabwe, and northern South Africa in 2001–04 [b](Reason *et al*., 2006).

The seasonal rainfall variations also showed a similar declining trend as annual rainfall; with autumn, winter and summer rainfalls being especially responsible for the sharp 1980s and 1990s declines. Richard *et al*. (2001) reported the Southern African sub-continent to have experienced severe droughts in the 1980s and beginning of the 1990s; with the magnitude of inter-annual summer rainfall series showing significant changes. Hoffman *et al*. (2009) have predicted the frequency of droughts to increase over the next hundred years in the winter rainfall regions of South Africa. Philippon *et al*. (2011) observed a link between seasonal rainfall variability and the El Niño Southern Oscillation (ENSO) cycle, with summer rainfall in Southern Africa affected by drier than normal conditions during El Niño and wetter than normal conditions during La Niña events. They reported South Africa winter rainfall (for the period 1979-2005) to have a positive correlation with the Niño 3.4 index that became significant since the 1976/77 climate shift. They explained that the high winter rainfall amounts recorded during the El Niño events were a result of longer and more frequent wet spells in Cape Town Region; while the low rainfall during the La Niña period were due to shorter, less frequent wet spells (Philippon *et al*., 2011). Again, not all El Niño events have similar effects. The 1997/98 event which had a weaker impact on southern African summer rainfall than the 1982/83 event; (Lyon and Mason, 2009) was responsible for strong winter rainfall anomalies over Western Cape (Philippon *et al*., 2011). In contrast, the 1991/92 event that had a very strong impact on summer rainfall over Southern Africa had a weak effect on winter rainfall anomalies over the Western Cape region (Rouault and Richard, 2005).

3.5.2 LUC change trends

It can be observed from the LUC classification that there was a significant reduction in the area occupied by intact vegetation, farms and some of the sand dune area between 1989 and 1999. The degraded and transformed vegetation invaded into most of the other LUC classes resulting in a substantial increase in these two classes during this sub-period. Dean *et al*., (1995) explained that bare areas within farms were associated with vegetation clearing, ploughing, localized overgrazing and areas where animals congregate such as water points, holding camps and feed lots. Large browsers like elephant, rhino and kudu as well as domestic animals like goats have historically played a significant role in limiting the spread of thicket vegetation. Generally, the study area was dominated by transformed vegetation (over 30 % of the land) with this proportion being highest in 1999 and lowest in 1989. The proportion of degraded land has increased over the years, from a low value of only 9 % in 1989 to levels as high as 25.3 % in 1999. Though there was a decrease in degraded land area to 17 % in 2009, it was still substantially higher than that originally recorded in 1989. Lechmere-Oertel *et al*. (2005) reported that transformation and degradation of succulent thicket in response to grazing is widespread in South Africa resulting in extensive replacement of a dense closed-canopy thicket by a few remnant trees with an ephemeral field layer. Smith and Wilson (2002) highlighted that less than 5 % of thicket vegetation in EC occurs in formal conservation areas making most of it vulnerable to overgrazing by domestic stock, bush clearing for agriculture, urban development, coastal resort development and invasion by alien vegetation. The Department of Economic Development and Environmental Affairs, DEDEA (2009) also reported an increase in the proportion of degraded land by 33.6 % in Eastern Cape (more so in the communal than commercial farming areas) which they attributed to bush encroachment and alien species invasion. Intact vegetation appeared to have been restored to its original 22 % areal coverage by 2009 and there was a general decrease in bare areas over the years. DEDEA (2009) noted that there has been a 2 % (210 905ha) growth in area covered by plantation forest in EC since 2004. This could explain why the 1999-2009 overlay had the highest mean NDVI value indicating an increase in vegetation cover. Since no distinction was made between natural and planted vegetation, the increase in area occupied by intact vegetation could have included the plantation areas.

The whole 1989 to 2009 study period showed an overall 1.8 % loss in farmland; while more of the bare areas were filled up by transformed, intact and degraded vegetation. The decrease in bare areas over time could be a result of decreased farming activity over the years in both the commercial and communal areas; while reduction in farmland could be due to changes in land-use patterns. Smith and Wilson (2002) noted a shift from pastoralism to game ranching within the EC province by private landowners since the 1980s in an effort to diversify their operations. They reported a 2.5 % (116 500 km^2) increase in areas converted from stock farming to game post 1996. This has been supported by Craig *et al*. (2011) who cited significant conversions of commercial farms to game ranches either for hunting or ecotourism over the past 20 years.

The LUC change - rainfall analysis showed varied trends between annual rainfall and NDVI for the three study periods examined. The most significant relations were observed in 1999 that had the least rainfall and recorded the lowest NDVI mean, maximum and sum. The two Grahamstown stations recorded the most significant rainfall-NDVI trends as they had the lowest NDVIs in 1999 when rainfall was lowest, the highest NDVI in 1989 when rainfall was highest and moderate NDVIs in 2009 when rainfall was moderate. The contribution of rainfall to land-use change is very variable as plant cover can be affected by many other factors such as unsustainable grazing, heavy browsing, land clearance and bush encroachment as mentioned before. Lechmere-Oertel *et al*. (2005) explained that the pseudo-savanna of transformed thicket seems to be stable in the short to medium term (50 years), so long as the climate is stable. However, they further explained that canopy trees appear to be vulnerable to episodic droughts, that cause significant mortality in transformed compared to intact habitats. The rates of tree mortality in this pseudo-savanna exceed seedling recruitment causing this unstable state to represent a transition to a desert-like state dominated by annual grasses, forbs and dwarf shrubs (Vetter, 2009). Spriggs (year unknown) highlighted that reduction in plant cover of thicket communities through goat farming and overgrazing was more severe in the more arid xeric and Spekboom thicket as plants struggled to recover under limited moisture conditions. Moreover, seedling recruitment after a drought also relies on a source of propagules in the form of a seed bank and dispersal from surviving plants.

It is difficult to state with certainty the effect of rainfall variability on thicket species transformation. Milchunas and Lauenroth (1993) highlighted that over shorter time scales (of less than a decade); rainfall and nutrients among other abiotic factors were stronger determinants of vegetation cover changes than grazing effects. Data over longer time scales is required to detect long-term cover changes since inter-annual and cyclical rainfall variation can obscure longer-term cover trends (Hudak, 1999). Moreover vegetation change is spatially heterogeneous; thus some areas are more resilient to transformation than others. This could be due to limited access to these areas by herbivores for prolonged periods (e.g. annual grasslands or grazing areas far from permanent water). The dominant plant species could also be tolerant of heavy defoliation (e.g. stoloniferous grasses), or nutrients, water and plant propagules lost from degrading patches may also be deposited and concentrated elsewhere in the landscape resulting in little net loss in such areas (Vetter, 2009). Thus the effects of vegetation transformation on secondary production tend to be masked by this spatial heterogeneity and periods of favourable rainfall. Fynn and O'Connor (2001) observed that despite rainfall having an inter-annual coefficient of variation of 30% in a semi-arid rangeland, there was evidence of grazing-induced changes in herbaceous composition and productivity in a drought year when the carrying capacity of the land was at its lowest. The changes in composition and productivity were attributed to low plant cover and elevated soil erosion under degradation.

3.6 Conclusion

The present study revealed that annual rainfall variability has shown a significant declining trend for three stations, namely Grahamstown, Uitenhage and Bathurst over the years. The overall analysis of pooled data for all stations showed a declining trend in annual rainfall; with decreases particularly significant in the 1980s and 1990s sub-periods. There was evidence from literature that explained the influence of the ENSO effect and sea surface temperature anomalies on annual rainfall variability, with below-normal rainfall generally observed during warm and above-normal rainfall generally observed during cold SST episodes. Thus some of the 1970s years that had above normal rainfall were due to the La Nina effect (or cold SST anomalies) while some 1980s and 1990s years that experienced below normal rainfall were as a result of the El Niño effect (or warm SST anomalies). Extreme wet events of DI and the daily rainfall subclasses of 10 mm and above closely followed the declining trend of annual rainfall for most stations over the years; both in terms of frequency and size of daily events. The frequency and length of dry spells on the other hand did not show any significant relations with annual rainfall or DI, with the length of dry spell only showing an increasing trend at Bathurst. Autumn, winter and summer rainfalls closely followed the trend of annual rainfall thereby contributing significantly to the sharp rainfall decline of the 1980s and 1990s sub-periods. There is evidence from other studies which shows that inter-annual rainfall variability over southern Africa has increased since the late 1960s and that droughts have become more intense and widespread in the region. Such alternating patterns of above-normal or below-normal rainfall periods make planning difficult for most governments. The impacts of extreme rainfall events on agriculture, engineering structures such as dams, water resource management and human livelihood are considerable, thus the possibility of long-term changes in the intensity of extreme events are of concern. Given that the EC Provinces experiences episodes of limited freshwater supply, a declining trend in rainfall in this area is undesirable. Since climate shifts are also linked to human activities such as fossil fuel burning and land-use change that cause unsustainable greenhouse emissions, it calls for more stringent measures to be implemented that curb extreme weather events like droughts and floods through disaster avoidance efforts. Increased public awareness and engagement with policy makers on the extent and consequences of climate change will help foster commitment to mitigation and adaptation measures. In such a scenario, conservation techniques such as afforestation, controlled grazing and protection of natural vegetation biomes could go a long way in minimizing greenhouse emissions and thus avoid major climate shifts. Approaches to mitigate climate shifts should involve taping into the indigenous knowledge systems of vulnerable groups e.g. smallholder farmers (since they depend on rain-fed agriculture) that use drought resistant crop varieties and other methods of copying with weather extremities. Municipalities could also direct resources towards preserving water and improving its quality by minimizing pollution of water bodies from urban and industrial wastes.

Land use and cover changes cannot be attributed to one factor alone but are a result of a combination of many factors. Human activity through cultivation, animal grazing-browsing and construction is undoubtedly a major contributor to land-use change. The main LUC changes observed were that land occupied by farms has decreased over the years, with most of it being converted to game ranches. Most of the changes were observed in the vicinity and west of Amakhala reserve. Most LUC changes were observed during the 1989-1999 sub-period. It was concluded that the study area is dominated by over 30 % of transformed vegetation land. The proportion of degraded land has also increased over the years, from a low value of only 9 % (76 000 ha) in 1989 to 17.5 % (144 743) in 2009. The increase in proportion of degraded land is undesirable as it only serves to exacerbate the declining rainfall trend observed in the study area. More conservation efforts should be promoted particularly in the communal areas since they have the greatest proportion of thicket vegetation that is unprotected. Controlled grazing should be introduced that gives vegetation enough resting time to allow for recuperation. More land could be turned into conservancies where animal numbers are controlled to avoid destruction by grazing and browsing. Within the conservancy areas themselves, the stocking densities of game animals must be heavily monitored since an uncontrolled population of elephant and rhino in particular have been known to be very destructive on thicket species. On the whole, the chapter supplied good information on the general rainfall trends and land-use cover changes that have been occurring in the Albany region. As such, more detailed information on actual amounts of carbon stocks and soil CO_2 emissions in the different cover types and environmental conditions can now be elucidated in subsequent chapters.

CHAPTER 4: PLANT AND SOIL CARBON SEQUESTRATION IN DIFFERENT LAND COVER TYPES

Having identified the major land-use and cover types in the Albany thicket region, this chapter now zones in on the second objective which is to quantify plant and soil C sequestered under different vegetation covers namely thicket, grassland and degraded land. It begins by giving some background information on thicket vegetation i.e. its growth form, diversity, factors affecting its growth and the impact of management practices on its stability. Later the chapter brings to light the actual amounts of carbon sequestered under thicket at various levels of degradation in comparison with grassland vegetation.

4.1 Introduction

The succulent Albany thicket is a unique vegetation type located in the semi-arid Eastern Cape Province of South Africa. It grows in hot dry valleys that receive between 200 and 450 mm annual rainfall; thus it is adapted to areas where soil moisture is limited for extended periods (Lechmere-Oertel, 2003). In terms of growth form, thicket vegetation is dense, perennial, semi-succulent, thorny and about 2–3m high, with a high diversity of plant species (Everard, 1987). It has a rich flora of about 1 600 species (Vlok *et al*., 2003), characterized by an assemblage of evergreen shrubs, tall succulents, deciduous and semi-deciduous woody shrubs, lianes, herbs, geophytes, some climbers, but very little grass (Palmer, 2004; Knight and Cowling, 2006). Cowling *et al*. (2005) described thicket as a dense, evergreen formation of weakly deciduous shrubs and low trees that are often spiny and festooned with vines. Albany thicket is restricted to deep, well-drained, fertile sandy loams with the densest thickets occurring on the deepest soils (Cowling 1984).

Much research has been done on the potential of thicket to sequester large reserves of carbon. Pierce *et al*. (2008) revealed that intact spekboom thicket stored an excess of 20 kg of carbon per square metre of vegetation in the arid south-eastern Cape. Mills *et al*. (2005) also observed that the concentration of C in soils of intact thicket was quite high for a semi-arid region, averaging 5.2% in the top 10 cm. This is because litter fall in succulent thicket is so high that it even compares with some temperate forests (Lechmere-Oertel, 2003). The high canopy cover in thicket lowers soil and air temperatures as well as light penetration thereby increasing the litter depth (Weatherall-Thomas, 2009). It is for this reason that efforts have been made towards its preservation and restoration.

Subtropical thicket ecosystems are generally stable, but they also have poor levels of resilience (Kerley *et al*., 1995). This is partly because of their poor seedling recruitment and survival. Their seedlings occur at low densities, and are limited to the microclimate underneath the canopy dominated by herbs and grasses; which restricts the regeneration of thicket species (Weatherall-Thomas, 2009). Where canopy cover is intact, degradation is limited. Canopy cover loss however, leads to loss of phytomass, species diversity and ultimately landscape carbon (Lechmere-Oertel *et al*., 2005). The loss of vegetation cover is followed by litter and soil loss. Soils in degraded subtropical thickets thus have reduced water infiltration and a tendency to crust (Mills and Fey, 2004). Canopy trees species in thicket vegetation e.g., *Pappea capensis*, *Schotia afra*, and *Boscia oleoides* tend to succumb to this altered microclimate and disappear from the landscape (Lechmere-Oertel, 2003). Palmer (2004) discovered that once damaged; the woody thicket species did not recover in time due to their slow growth rates. The main forage species *Portulacaria afra* has a slow recovery period of up to 18 months after 50% defoliation by goats (Mills *et al*., 2005). This slow growth rate accentuates the need to preserve thicket plants in situ.

Unfortunately, prolonged domestic herbivory in succulent thicket has led to significant changes in vegetation cover, structure and diversity of endemic geophytes and succulents, leaving behind a ‘pseudosavanna’ that is dominated by ephemeral perennial grasses and dwarf karroid shrubs (Kerley et al., 1995). Heavy goat browsing can transform the dense closed-canopy shrub-land into an open community comprising scattered and degraded thicket clumps and isolated trees in a matrix of ephemeral herbs (Lechmere-Oertel *et al.*, 2005). Here, the canopy of the perennial patches is opened by the livestock, exposing the litter layer and soil surface to dispersion. Powell (2009) reported that excessive browsing of *Portulacaria afra* dominated thicket by goat reduces natural species diversity, above- and below-ground carbon stocks, soil quality (Mills and Fey, 2004), plant productivity and ultimately livestock or game stocking capacity. Mills *et al*., (2005) reported semi-arid thicket transformation by goat pastoralism to have resulted in a combined loss of up to 82 t C ha^{-1} from soil and plant biomass.

It is against this background that this chapter aims to quantify plant and soil C sequestered under thicket, grassland and degraded vegetation. It served to provide background knowledge on the chemical and physical soil properties at the study site as well as to assess the carbon sequestration potential of the different vegetation covers. These same sites were later used to monitor CO_2 effluxes and for carbon isotope measurements.

4.2 Study area

This study was carried out at Amakhala Game Reserve; located in the Greater Addo and Frontier Country area, in the Eastern Cape (EC) Province of South Africa (Figure 4.1). It is situated at the end of the Garden route; about 90 km North East of the city of Port Elizabeth, on a 7500ha plot. It inhabits a variety of animals such as lion, elephant, rhino, buffalo, cheetah, giraffe, zebra, wildebeest and plenty of antelope species. The Amakhala Game Reserve began in 1999 as a joint conservation venture that converted ranch and farming land into game reserve land. Historically, the area comprises several vegetation biomes i.e. savanna, Albany thicket, grassland and Karoo vegetation (Figure 4.2). The long history of goat farming as a major land-use has transformed this area to open grasslands with isolated patches of thicket and blue bush (*Pteronia incana*) invasion on most of its degraded land (Odindi, 2009). The area is semi-arid; and rain falls throughout the year with annual ranges of 380-570 mm. Summer temperatures are in the ranges of 16-30^{o}C while winter temperatures are 5-22^{o}C (Odindi, 2009). The reserve soils are predominantly sands and clays derived from sandstone and shale of the Algoa formation (Odindi and Kakembo, 2009).

Figure 4.2 shows the various vegetation biomes at Amakhala reserve. This study focused on carbon sequestration potential of thicket and grassland vegetation only as it was not possible to assess all biomes. Ultimately, three sites were chosen which comprised an intact thicket, a degraded thicket and a grassland site for comparison. These same sites were used to characterize plant and soil carbon sequestered, to analyse variations in 12/13 C isotope ratios and to monitor soil CO_2 effluxes over time.

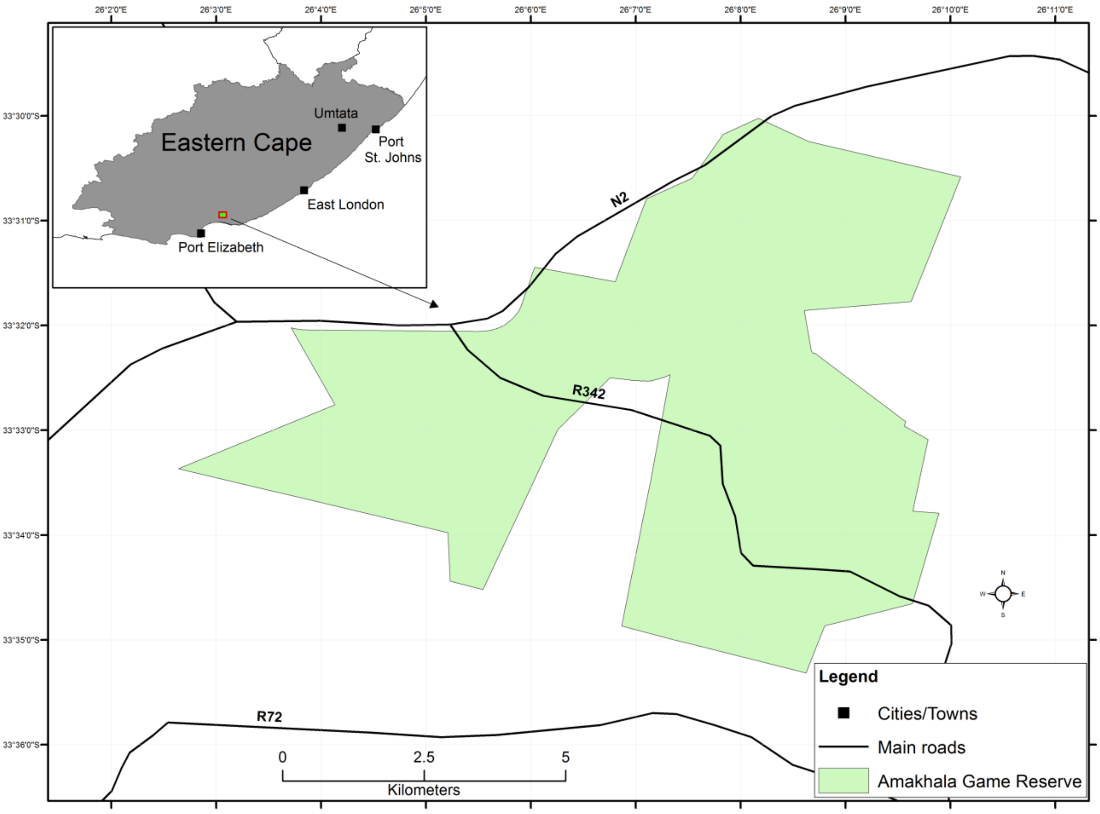


Figure 4.1: Map showing the location of Amakhala game reserve.

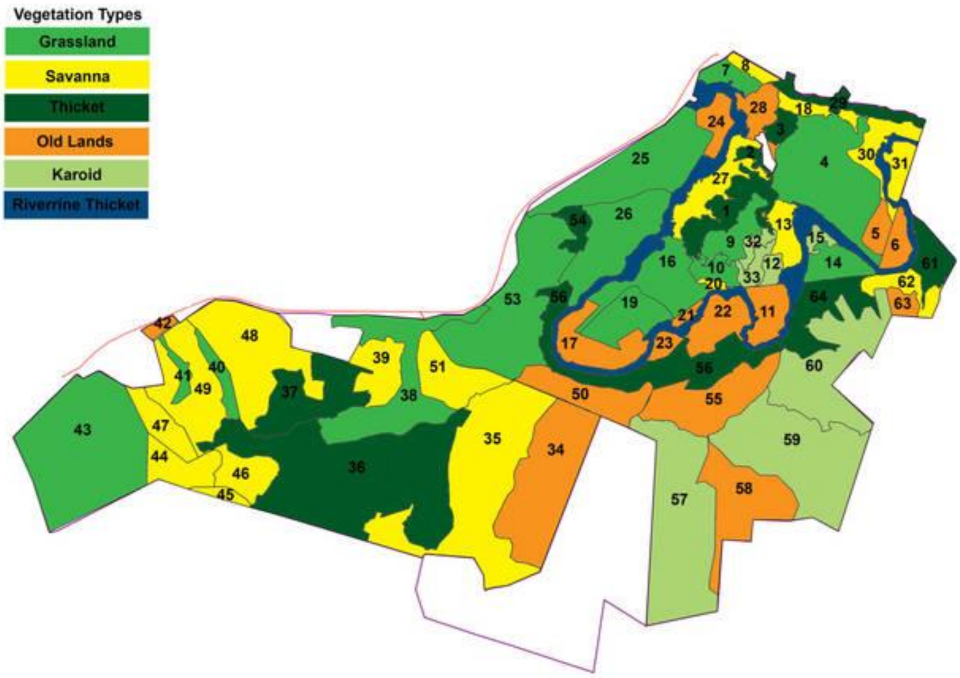


Figure 4.2: Vegetation types at Amakhala game reserve

*The numbers denote plot demarcations used for identification purposes by the reserve

4.3 Methodology

4.3.1 Soil characterization

Each land cover (i.e. intact thicket, degraded thicket and grassland) had a total size area of about ten hectare (500 x 200 m) and had one transect that traversed it diagonally with five points selected at regular intervals along it that were later used as sampling points for gas flux measurements (Figure 4.3). Five replicates were used per site with the points set up in a completely randomized design blocked according to slope. Soil samples were collected randomly around each gas monitoring point from each biome at depths of 0 – 10, 10 – 20 and 20 - 30 cm. A total of five replicates were used per each depth. The samples were analysed in the laboratory for organic C (using the modified Walkey - Black method), total N (Kjeldahl method), texture (hydrometer method), pH in water and 0.01M $CaCl_2$ (pH meter), bulk density (core method), and the exchangeable bases calcium (Ca), magnesium (Mg), sodium (Na) and potassium (K) (using the Ammoniun acetate method) (Anderson and Ingram, 1989). The same samples were later used to assess 12/13 C isotope ratios in soil organic matter that will be detailed in Chapter 5.

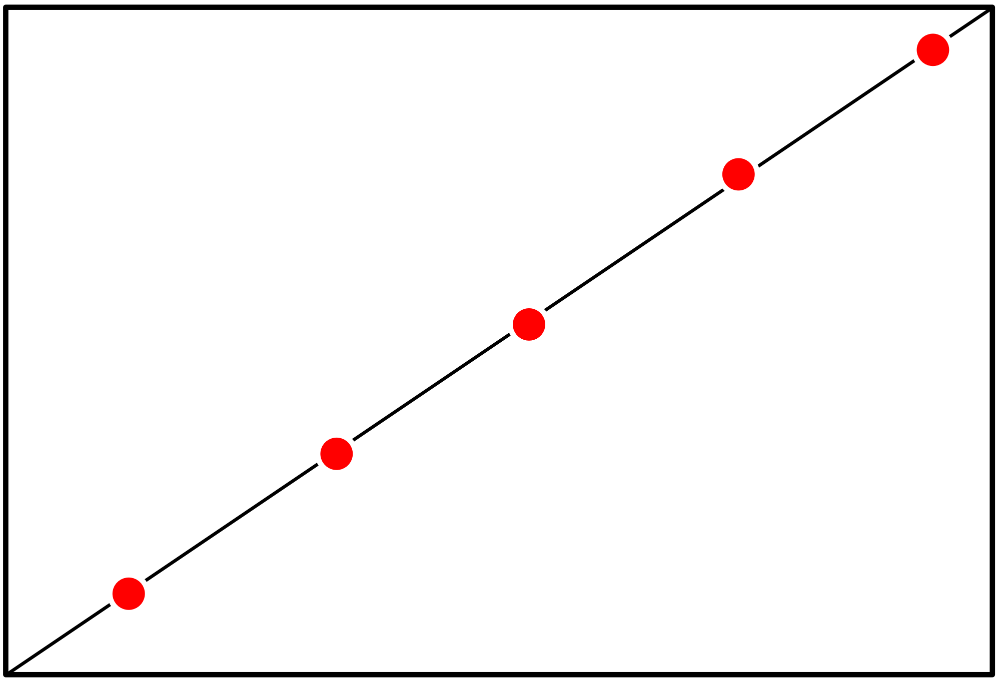


Figure 4.3: Diagram showing the gas flux monitoring points for each land cover

4.3.2 Species inventory

In order to do plant assessments, the rectangular area shown in Figure 4.3 above was further sub-divided into three quadrats that had the most representative vegetation of the site and were used as replicates for each land cover. These quadrats had a total size area of 50 x 20 m each. It was in these three quadrats that aboveground biomass measurements were done. At the intact thicket sites, canopy trees and woody shrubs were counted and biomass measurements were estimated in the main quadrat; while smaller plants i.e. grasses, forbs and dwarf shrubs were assessed in smaller sub-quadrats of 1 x 1 m that were within the main quadrats. A total of three sub-quadrats were used in each main quadrat. There were no canopy trees or woody shrubs at the degraded thicket and grassland site. Thus smaller quadrats of 10 x 10 m were used at these sites, with a total of three quadrats per site. The five gas monitoring points were all located within the three quadrats at each site. The initial stage of the plant assessment was conducted to make an inventory of the plant species within each land cover. This was done by identifying the most dominant species; that were then used as a guide to draw up the most representative quadrats of each area. The dominant species identified were then put into categories (guilds) based on their growth forms.

4.3.3 Aboveground biomass and plant carbon assessment

Plant aboveground biomass measurements in intact thicket were done for the dominant species identified. This was achieved by using allometric equations (Table 4.1). The initial idea was to calibrate the equations but it was not possible as the study area was a conservation reserve in which destructive plant sampling was not permitted. Thus pre-existing allometric equations that were derived by other researchers were used. Most of the equations were derived from the work done by Lessings (2007) and Powell (2009) on thicket vegetation. Key parameters that were considered in the plant measurements were plant height, canopy dimensions (height and diameter), basal stem diameter and stem numbers. The mean canopy area (CA) was derived from the length of the widest axis (W) and the axis perpendicular to it (L) using the following formula:

$$CA = \pi\, W * L / 4$$

The plant total basal area (BA) was calculated as:

$$BA = \sum (\pi / 4 \times DBH^2)$$

DBH = Diameter at breast height of each stem

*The equation for BA takes into account the total stem numbers

(The above equations were obtained from Uzoh and Ritchie, 1996)

The allometric equations that were used to calculate dry mass of the canopy trees and shrubs were developed by Lessings (2007) as follows:

Table 4.1: Biomass equation for some thicket species

Plant species	**Allometric equation**
Portulacaria afra	$Y = 2.71x - 1.80$
Schotia afra	$Y = 2.84x - 1.47$
Euclea undulata	$Y = 2.55x - 0.96$
Azima tetracantha	$Y = 2.88x - 1.22$
Carissa haematocarpa	$Y = 2.51x - 0.90$
Gymnosporia polyacantha	$Y = 2.97x - 1.18$

Where Y = Log Dry mass (grams) and X = Log Stem diameter (mm)

*Dry mass is the total mass of leaves, stems and branches.

To estimate the biomass C, the following allometric equations (Table 4.2) were used as derived from Powell (2009).

Table 4.2: Biomass C equations for some thicket species

Plant species	**Allometric equation**
Portulacaria afra	$\text{Log}Y = 1.1043 \text{ Log BA} + 2.4464$
Rhus longispina	$\text{Log}Y = 1.1012 \text{ Log CA} - 0.2938$

Where Y = biomass C (kg), BA is basal area (m^2) and CA is canopy area (m^2)

The biomass C for all the other species was estimated using the general dry mass to C conversions factor of 1: 0.48 (Lessings, 2007).

Plant aboveground biomass estimates for the degraded thicket and grassland was different from that of the intact thicket trees and shrubs since these 2 sites only comprised herbs and grasses. It involved biomass assessment of the dominant species identified from smaller 1 x 1 m sub-quadrats within the main quadrat.

4.3.4 Statistical analysis

A two-way analysis of variance (ANOVA) was done for soil measured variables, above-ground biomass and plant C to assess variations between treatment means. Treatments for the soil analyses included the land cover and soil depth and these were blocked against the sampling point; while plant variables were analysed and compared for each land cover type. The analysis was done using the statistical package GenStat 14.1 for Windows.

4.4 Results

4.4.1 Soil properties

Tables 4.3 and 4.4 present summaries of the soil analysis results. The results showed that soil pH had moderate ranges of 5.7 to 7.5 pH water, that did not different between the three cover types ($p = 0.285$). It however increased with depth especially at the intact thicket (IT) and grassland (G) ($p = 0.03$). Soil texture varied from sandy loam to sandy clay loam across sites. The grassland in particular was quite sandy (> 70 % sand), while the degraded thicket (DT) had substantial amounts of silt averaging 20 % ($p < 0.001$). Exchangeable bases generally increased with depth across sites, with the IT recording relatively high levels of Ca, Mg and K compared to the other two sites. There were low levels of Na (< 0.6 $\text{Cmol}_c\ \text{kg}^{-1}$) at all sites. The calcium (Ca): Magnesium (Mg) ratio ranged from 2 to 7. The G recorded the lowest amounts of all the bases ($p < 0.001$).

Table 4.3: Soil properties of the study sites

Property	Depth (cm)	Intact Thicket	Degraded Thicket	Grassland
pH in Water	0-10	5.7^{a}	6.3^{b}	6.4^{b}
	10-20	6.5^{b}	6.3^{b}	6.6^{b}
	20-30	7.5^{c}	6.3^{b}	6.8^{d}
pH in $CaCl_2$	0-10	5^{a}	5.2^{b}	5.5^{b}
	10-20	5.7^{b}	5.3^{b}	5.7^{b}
	20-30	6.7^{c}	5.4^{b}	5.9^{d}
Texture	0-10	$(69: 14 : 17)^{a}$ Sandy Loam	$(66: 14 : 20)^{b}$ Sandy Loam	$(73.5: 10.5 : 16)^{d}$ Sandy Loam
	10-20	$(64.5: 15.5 : 20)^{b}$ Sandy Loam-Sandy Clay Loam	$(64: 15 : 21)^{b}$ Sandy Loam-Sandy Clay Loam	$(72.5: 12.5 : 15)^{d}$ Sandy Loam
	20-30	$(66.5: 13.5 : 20)^{a}$ Sandy Loam	$(61.5: 18.5 : 20)^{c}$ Sandy Loam-Sandy Clay Loam	$(69.5: 17 : 13.5)^{e}$ Sandy Loam
Exch Calcium ($Cmol_ckg^{-1}$)	0-10	25.6^{a}	7.2^{c}	4.6^{c}
	10-20	22.6^{a}	7.3^{c}	4.6^{c}
	20-30	31.3^{b}	7.6^{c}	6.3^{c}
Exch Magnesium ($Cmol_ckg^{-1}$)	0-10	3.6^{a}	2.7^{ab}	1.1^{c}
	10-20	3.9^{a}	2.9^{ab}	1.2^{c}
	20-30	4.4^{a}	3.3^{ab}	1.9^{bc}
Exch Sodium ($Cmol_ckg^{-1}$)	0-10	0.3^{a}	0.3^{a}	0.2^{d}
	10-20	0.4^{b}	0.4^{b}	0.2^{d}
	20-30	0.5^{c}	0.4^{b}	0.3^{a}
Exch Potassium ($Cmol_ckg^{-1}$)	0-10	1.4^{a}	1.4^{a}	0.8^{da}
	10-20	1.8^{b}	1.3^{a}	1.0^{a}
	20-30	2.4^{c}	1.1^{a}	1.3^{a}

Values in brackets in the texture column show % Sand: Clay: Silt

*Values with the same superscript are not significantly different; those with different superscripts are statistically different for each depth – land cover treatment per measured parameter.

Table 4.4 Summary statistics for soil properties

Variable	Land cover p value	Depth p value	Land cover x Depth interaction
pH water	0.285 (0.2)	0.003** (0.2)	0.017* (0.3)
% Clay	0.088 (1.1)	0.012* (1.1)	0.075 (1.9)
% Silt	< 0.001*** (1.4)	0.738 (1.4)	0.584 (2.4)
% Sand	< 0.001*** (1.5)	0.055 (1.5)	0.666 (2.6)
Ca	< 0.001*** (17.8)	0.263 (17.8)	0.603 (30.9)
Mg	< 0.001*** (2.4)	0.346 (2.4)	0.998 (4.2)
Na	< 0.001*** (0.4)	0.035* (0.4)	0.867 (0.7)
K	< 0.001*** (2.5)	0.034* (2.5)	0.055 (4.4)
Organic C	< 0.001*** (0.1)	< 0.001*** (0.1)	< 0.001*** (0.2)
Total N	< 0.001*** (0.008)	< 0.001*** (0.008)	< 0.001*** (0.01)

***Very highly significant (< 0.001); **highly significant (< 0.01), *Significant (< 0.05)
**Values in brackets represent the standard error differences of means*

4.4.2 Variations in soil organic carbon (C) and nitrogen (N) across study sites

Organic C was quite high in intact thicket, reaching levels of 3.4 % (i.e. 3.4 t C / ha of land) in the top 10 cm then decreasing with depth ($p < 0.001$; Table 4.3 and 4.5). There were moderate levels of organic C at DT site (1.1-1.3 %) and very low C in the grassland (≤ 0.5 % C). Total N followed more or less the same trend; with N values being low at the grassland site ($p < 0.001$; Table 4.3 and 4.5). This resulted in lower organic C: N values at the grassland and moderate ratios in the degraded and intact thicket sites.

Table 4.5: Soil organic C and total N across cover types

Property	Depth (cm)	Intact Thicket	Degraded Thicket	Grassland
Organic Carbon (%)	0-10	3.4^a	1.3^d	0.5^f
	10-20	2.1^b	1.2^e	0.4^f
	20-30	1.7^c	1.1^e	0.4^f
Total N (%)	0-10	0.24^a	0.11^d	0.06^f
	10-20	0.14^b	0.09^e	0.05^f
	20-30	0.12^c	0.08^e	0.05^f
Organic C: N ratio	0-10	13.3^a	11.8^d	8.3^f
	10-20	15^b	13.3^e	8^f
	20-30	14.2^c	13.8^e	8^f

*Values with the same superscript are not significantly different; those with different superscripts are statistically different for each depth – land cover treatment per measured parameter.

4.4.3 Variation in vegetation at study sites

Vegetation at the IT site was characterized by a dense thicket with a diversity of growth forms (Table 4.6). Species varied from the woody trees *Euclea undulata* and *Schotia afra,* to multi-stemmed woody shrubs such as *Rhus longispina, Sideroxylon inerme, Gymnosporia* species; *Carissa haematocarpa, Azima tetracantha* and the popular succulent shrub *Portulacaria afra* (spekboom). The under-storey of this biome also had a diversity of dwarf succulent shrubs and forbs. These different species were put into guilds based on their growth forms which included canopy trees, woody shrubs, succulent shrubs and ephemerals. Table 4.6 gives a detail of the growth forms.

There were no woody species left in the degraded thicket; but it had been heavily invaded by the alien species *Pteronia incana* (blue bush), grasses such as *Themeda triandra* and *Panicum maximum* and the herbs *Chrysocoma ciliate, Crassula mesembryanthoides, Senecio linifolius* and *Cyphia sylvatica*.

Table 4.6: The main plant guilds identified in the intact thicket at Amakhala

Guild	Characteristics	Dominant species
Canopy trees	Stemmed woody canopy plants up to 5 m in height, primarily deciduous and drought tolerant.	*Euclea undulata* *Schotia afra* *Sideroxylon inerme*
Woody shrubs	Multi-stemmed woody plants, up to 4 m tall, often with sharp branchlets, spines or thorns, deciduous and drought tolerant.	*Gymnosporia polyacantha* *Azima tetracantha* *Rhus longispina*
Succulent shrubs	Multi-stemmed semi-woody shrubs, up to 3 m tall, have evergreen succulent leaves (mainly CAM photosynthesis).	*Portulacaria afra* *Carissa haematocarpa*
Herbaceous species	Short-lived perennial grasses, forbs and herbs (sometimes succulent) that closely track rainfall, perennial karroid dwarf shrubs.	*Crassula expansa* *Crassula mesembranthoides* *Senecio species* *Phyllobolus splendens* *Setaria sphacelata*

The Grassland site had a predominance of *Cyanodon dactylon* grasses, a bit of *Panicum maximum* as well as herbs such as *Senecio linifolius* and *Chrysocoma* and some *Crassula*, *Atriplex*, *Fabaceae*, *Rubus* and *Commelina* species.

Above ground biomass in intact thicket was quite high especially for the succulent shrub *Portulacaria afra* and the canopy trees *Euclea undulate, Rhus longispina* and *Schotia afra* (Table 4.7). It reached values as high as 330 000 kg/ha in intact thicket; but was only 22 000 kg/ha in degraded thicket and as low as 6 700 kg/ha in grassland vegetation (p = 0.01). The biomass carbon also followed this trend (i.e. 158 000, 10 600 and 3 200 kg/ha respectively). This means the conversion of IT to a DT would result in a net loss of 147 400 kg/ha of biomass C. Soil organic C density ranged from 24 000 kg/ha in IT, 12 000 kg/ha in DT to only 4 500 kg/ha in grassland. Again restoring DT to the IT condition would result in a net gain of 12 000 kg/ha in soil organic C. Thus there was considerable reduction in above-ground biomass, plant and soil organic carbon at the degraded thicket and grassland site (Figure 4.4). The bulk of the C was stored in above-ground biomass at all sites.

Table 4.7: Variation in plant parameters of Intact Thicket

Species	**Average Plant Number (0.1 ha)**	**Height (m)**	**Canopy area (m^2)**	**Basal area (m^2)**	**Stem numbers**	**Above-ground Biomass (kg/ha)**	**Biomass C (kg/ha)**
Euclea undulata	16	3.3 – 4.1	3.96 – 9.85	0.33 – 1.74	1 - 4	12 283	5 895
Sideroxylon inerme	5	2.4 - 3	2.04 – 3.87	0.1	1	836	401
Schotia afra	12	2.2 – 5.2	1.45 – 12.91	0.19 – 1.35	1 - 3	4 858	2 331
Gymnosporia polyacantha	17	1.6 – 2.8	0.55 – 4.81	0.01 – 0.06	1 - 12	2 805	1 346
Azima tetracantha	35	1.5 – 2.3	0.57 – 1.99	0.02 – 0.11	1 - 4	1 567	752
Rhus longispina	11	2.2 – 4.5	1.56 – 7.76	0.05 – 0.78	1 - 6	14 928	7 165
Portulacaria afra	24	1.8 – 3.9	1.33 – 6.33	0.12 – 1.59	2 - 6	80 375	38 580
Carissa haematocarpa	10	1.5 – 2.6	0.88 – 4.49	0.01 – 0.48	1 - 5	996	478

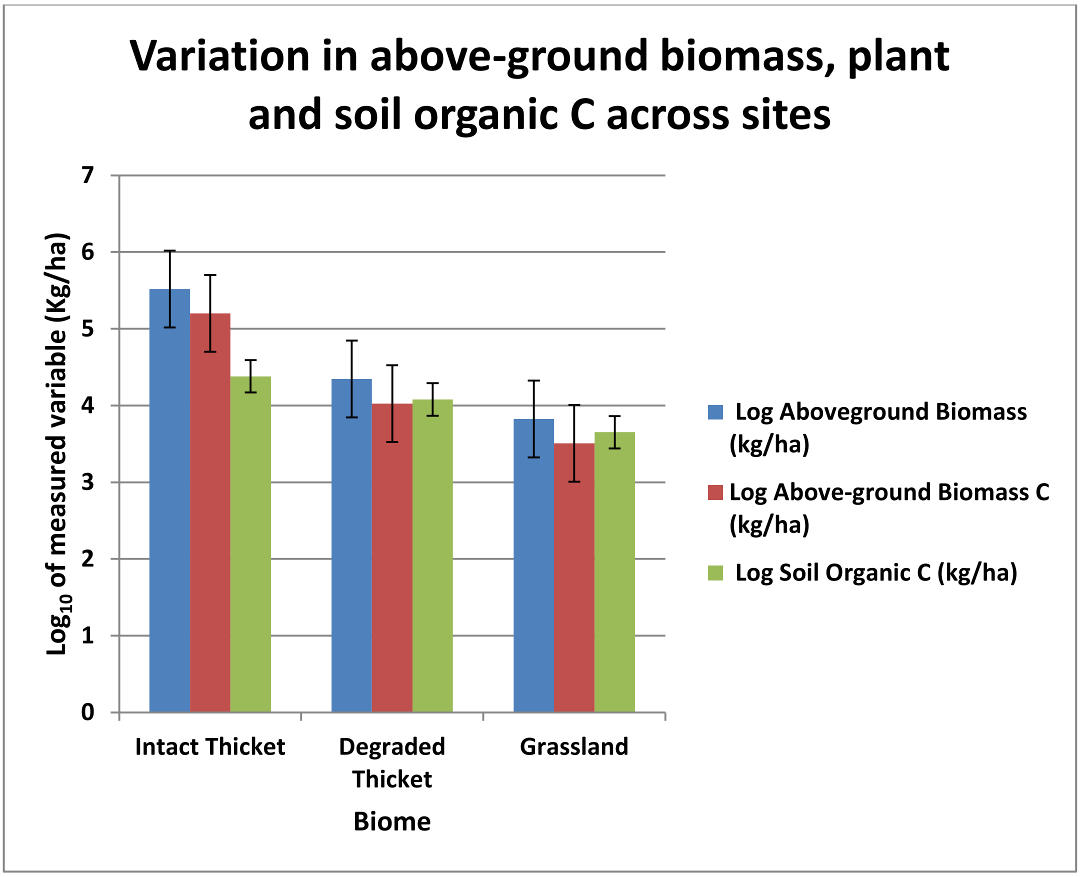


Figure 4.4: Comparison of above-ground biomass, plant and soil C density across sites

**Values for soil organic C were converted to mass per unit area basis by factoring in the bulk density and sampling depth (i.e. 10 cm).*

4.5 Discussion

4.5.1 Species distribution

This study confirmed the high floristic diversity of thicket vegetation in an undisturbed state. It was characterised by rich growth forms of canopy trees, woody shrubs, succulent shrubs and ephemerals most of which were completely eliminated under degraded thicket. The disappearance of succulent thicket species upon degradation was significant; resulting in the proliferation of ephemerals and invader species such as *Pteronia incana* (blue bush). Lechmere-Oertel, (2003) stated that the transformation of thicket results in replacement of large areas of dense thicket with a ‘pseudo-savanna’ of remnant canopy trees as well as ephemeral short- lived perennial grasses and forbs. The author further explained that the succulent species regenerate slowly after disturbance and are eliminated if these disturbances occur at intervals of less than a few decades. In our study; the canopy species had been totally eliminated leaving behind weak ephemerals with the invader species *Pteronia incana* dominating this land cover. Wildlife, particularly elephant and rhino contributed a lot to the destruction, since succulent plants form a great part of their diet. Pierce *et al*. (2008) also noted that large browsers like elephant (*Loxodonta africana*), rhinoceros (*Diceros bicornis, Ceratotherium simum*) and kudu (*Tragelaphus strepsiceros*) played a significant role in transforming succulent thicket. Vetter (2009) explained that the rate of tree mortality in the pseudo-savanna created after thicket transformation exceeded the system's seed recruitment capacity; resulting in the creation of an unstable desert-like system dominated by annual grasses, forbs and dwarf shrubs. Vlok *et al*. (2003) reiterated the need to avoid succulent thicket transformation as their restoration does not occur spontaneously, even with complete resting from herbivory.

4.5.2 Above-ground and C biomass

The study also showed that degradation of intact thicket through animal browsing led to a marked reduction in above ground biomass and C of thicket species. This conversion of IT to a degraded condition led to a net loss of 147 000 Kg of biomass C / ha. Mills *et al*. (2005) observed values of 160 000 kg/ha of dry matter in living biomass and surface litter of an intact thicket which was drastically reduced under transformed thicket; translating to an average biomass C loss of up to 58 000 kg/ha as a result of goat pastoralism under transformed thicket. Lechmere - Oertel *et al*. (2005) also found a difference in total biomass between intact and transformed thicket of 80 000 kg/ha. Significant amounts of biomass loss could lead to a reduction in overall functionality of the land as soil surfaces are exposed, causing problems such as soil dispersion, surface crusting which in turn lead to reduced infiltration, accelerated runoff and erosion (Lechmere - Oertel *et al*. 2005). The soils under degraded thicket were generally characterised by higher bulk densities (of mostly above 1 g cm^{-3}) and higher penetration resistance (Chapter 6) compared to intact thicket soils that were much softer and had densities of mostly below 1 g cm^{-3}. They also had up to 12 000 kg less organic C / ha of land than IT soils. All of this in turn reduces plant biomass production and consequently the animal stocking capacity of the land (Dean *et al*., 1995).

4.5.3 Status of soil properties at study sites

The results from soil analyses showed the IT to be a more favourable site as it was characterised by higher amounts of exchangeable bases, organic C and N; while the DT had moderate levels and the G had poor nutrient levels. Soil pH did not differ much between sites but showed a general increasing trend with depth. This might be explained by the corresponding increase in clay amounts with depth observed at the sites. Clay is usually transported through processes such as clay eluviation from the A horizon to the lower B horizon. In terms of texture, both thicket sites had favourable amounts of silt and clay compared to the grassland sites. Furthermore, the G generally had very low values of all measured soil and above-ground variables showing a poor nutritional status of its soil. This could explain its poor biomass accumulation. The G was characterized by very sandy soils (> 70 %) with low levels of exchangeable bases and nitrogen.

Results for soil organic C were not as high as those reported in similar thickets. Mills *et al*., (2005) reported over 5 % soil C in the top 10 cm; while in our case it was a little over 3%. Nevertheless, soil C under intact thicket was still substantially higher ($p < 0.001$) than that observed under grassland or degraded thicket. Again the DT site though degraded; seemed to perform better in terms of above-ground biomass and C accumulation than grassland. This showed that thicket vegetation had great potential to sequester carbon, more so if preserved in its pristine state. Given the increase in proportion of transformed and degraded land reported in the previous chapter, and the resultant decline in biomass production and C sequestration potential, destruction of thicket vegetation is not desirable and must be avoided. Preservation of thicket not only results in large reserves of C but also creates fertile soils that are protected from crusting, dispersion and erosion by the thick litter layer produced in such systems. The thick litter layer is also efficient at preserving moisture that can be used under extremely dry conditions when rainfall is marginal. Absence of this leaf litter under degradation exposes the soil to desiccation, such that the degraded plants struggle to recover due to moisture stress (this is illustrated in chapter 6). Moreover, the exposed soils are also prone to runoff and erosion which ultimately leads to siltation of surrounding freshwater bodies that are already failing to meet the increased water demand.

4.6 Conclusion

High biomass and carbon accumulation under undisturbed thicket was quite evident. This translated to soils under such vegetation also sequestering larger reserves of C. Disturbance of succulent thicket species resulted in a huge loss in phytomass (about 147000 Kg / ha biomass) and consequently ecosystem carbon (up to 12 000 Kg / ha). It also resulted in creation of an unfavourable soil environment that is exposed and prone to dispersion, crusting, erosion and accelerated runoff. This degradation will in-turn affect recovery of thicket species and leave behind invasive plant species as well as weak ephemerals. Since we observed in the previous chapter that the proportion of the area occupied by transformed and degraded thicket has been increasing over the years; and the resultant low quality environment created by such degradation, it is imperative to raise awareness on the need to avoid further degradation of thicket. Biomass and C accumulation under grassland was not good; and this was attributed to a poor nutritional level of the soil. It is recommended to maintain succulent thicket in its pristine condition as its restoration is difficult and does not occur spontaneously.

CHAPTER 5: ^{13}C AND ^{15}N ISOTOPE RATIOS AS INDICATORS OF LAND-USE CHANGE AND SOIL ORGANIC MATTER TURNOVER

The previous chapter looked at the different growth forms under thicket and the amount of biomass as well as carbon they sequester. This chapter now looks at the types of photosynthetic pathways of these species in comparison with the C and N isotropy of soil organic matter in the different land cover types. It gives a more detailed investigation to quantify the extent of the change in land use/cover under real field conditions since general land use cover change detection has already been done for the Albany thicket region in chapter 3. This chapter thus looks at the third objective which is to assess the influence of ^{13}C and ^{15}N isotope ratios as indicators of land-use change and soil C residence time. The focus is on quantifying the carbon isotope composition of soil organic matter under different vegetation covers; and then use it to assess vegetation as well as land-use changes that have occurred over the years at the study site. An additional experiment to quantify natural abundance of nitrogen ratios (i.e. δ^{15}N) was also included to complement this study by helping to elucidate on the nutrient cycling processes occurring in the soil at the study sites. Three land cover types namely intact thicket, degraded thicket and grassland at Amakhala reserve were used in the assessment.

5.1 Introduction

About 98.89 % of carbon occurring in nature is in the form of ^{12}C, while only 1.11 % is ^{13}C (Weaver *et al*., 1994). The relative proportions of these two stable isotopes will vary in different environmental components depending on the isotopic fractionation that occurs during various physical, chemical and biological processes. The ^{13}C / ^{12}C ratio (also known as the ^{13}C depletion; and denoted δ ^{13}C) defines the proportions of C13 and C12 isotopes of organic C in terrestrial ecosystems, and is largely determined by the isotopic fractionation that happens during photosynthesis (Weaver *et al*., 1994). Thus C_3 plants (i.e. those that utilise a Calvin–Benson photosynthetic pathway) will exhibit greater discrimination against ^{13}C than C_4 plants (i.e. those that utilize the Hatch–Slack photosynthetic cycle) (Muccio and Jackson, 2009). Plants with the Crassulacean acid metabolism (CAM) can switch between the C_3 and C_4 metabolism, depending on sunlight availability. The stable isotope ratios of elements in both organic and inorganic components of the plant–soil system often record information relating to the identity of the processes that formed those components; the rates of these processes and the environmental conditions prevailing at the time the components were formed (Boutton *et al*., 1998). Thus a wealth of pedological, biogeochemical and ecological information can be obtained from stable isotope analyses of soils. The ^{13}C depletion for plants generally varies from -20 to -8‰ for C_4; -34 to -10 ‰ for CAM; and -35 to -22 ‰ for C_3 plants (Muccio and Jackson, 2009).

Soil carbon isotopic signatures ($\delta\ ^{13}C$) are commonly used to reconstruct plant community history, determine sources of soil organic carbon (SOC), and quantify SOC turnover rates (Bai *et al*., 2012). They closely correspond with the $\delta\ ^{13}C$ of plant residues entering the system through litter-fall and root turnover. As plant residues are incorporated into the soil system, their $\delta\ ^{13}C$ values may be modified slightly from the original by isotope fractionation associated with microbial activity as well as by differential decay of isotopically unique soil organic matter compounds (Bai *et al*., 2012). Because of this, variation and evolution of soil $\delta\ ^{13}C$ values over time is controlled firstly by carbon inputs from vegetation and secondly by biological decay processes. Changes in the relative proportions of C_3, C_4 and CAM plants can denote the difference between the isotopic compositions of the current plant community and that of soil organic matter (Boutton *et al*., 1998). This difference will be largest soon after a change in vegetation, but will decrease over time as C from the previous plant community decays out of the organic carbon pool and is replaced by new carbon derived from the new plant community. As a result, the isotopic discrepancy created by the vegetation change will persist for some time determined by the soil organic matter turnover rate.

As discussed in the previous chapter, the thicket biome comprises mostly succulent plants. Succulent plants from arid regions are usually diverse in their CO_2 fixation patterns, as they switch between C_3 photosynthesis during the day and CAM metabolism at night (Mooney *et al*., 1977). The ability to fix significant amounts of CO_2 at night through CAM metabolism is regarded as an adaptive mechanism to conserve water in arid regimes. This is because with CAM, carbon dioxide is assimilated at night when potential evapotranspiration from the plant is lowest, thus the amount of water lost per unit of carbon gained is relatively low. It is therefore possible to infer how much CO_2 is fixed during the day and how much of it is fixed at night from the C isotope ratios of CAM plants. Farquhar *et al.* (1989) noted that upon exposure to increased drought, some plants could shift from C_3 to CAM; a mechanism that is reversible, depending on plant water availability, and is very common in leaf succulents of arid habitats. Studies by Mooney *et al*. (1977) also revealed that under natural environmental conditions; the larger succulents showed carbon isotope fractionation that was predominantly CAM, while the smaller leafy shrubs had more flexibility between C_3 and CAM photosynthesis. They explained that the small leafy succulents were more likely to undergo large shifts in their seasonal internal water supply; while massive stem succulents maintained solely night-time CO_2 fixation during periods of both abundant and limited soil moisture availability thereby making them more water-use efficient than the smaller succulents. Ehleringer *et al*. (1998) re-iterated that the ratio of photosynthesis to transpiration or that of biomass production to water consumption was higher in arid zone plants than in those from more mesic habitats. Not all succulent species are capable of CAM photosynthesis however. Comparison of the ^{13}C depletion between succulent plants at different levels of degradation or transformation will be a good indicator of the organic matter turnover as well as land-use changes that have occurred over time.

Although the use of natural abundance ratios of nitrogen (i.e. $\delta^{15}N$) is not as well established as C isotropy, a great deal can still be learned from comparing the ^{15}N among plants within an ecosystem, between plants and their source of N, and among plant components (Michener and Lajtha, 2007). The natural abundance of the rare stable isotope ^{15}N, relative to that of the more abundant ^{14}N (i.e. ^{15}N / ^{14}N) is utilised in such studies. The $\delta^{15}N$ of the atmosphere is 0 ‰; thus the $\delta^{15}N$ of plants utilising atmospheric N_2 should also be close to 0‰, (Robinson, 2001). However, there are other sources of N in the soil, such as ammonium (NH_4^+) and nitrate (NO_3^-) that contribute to the $\delta^{15}N$ of plants. Some key applications of $\delta^{15}N$ in plants include assessing contributions of symbiotic nitrogen fixation, atmospheric deposition, mycorrhizal infection and dissolved soil N to plant N uptake (Michener and Lajtha, 2007). Several problems that stem from the use of N isotropy include the huge diversity of N sources in the soil; with many plants showing distinct preferences to any of them. Furthermore, plants can change their preference for NO_3^- versus NH_4^+ depending on environmental conditions; such as the rate of nitrification or fertilizer application (Pardo *et al*., 2007). Thus the $\delta^{15}N$ of a system not only reflects the $\delta^{15}N$ of the N source, but also N isotope fractionations, gains, losses and N pool mixing making its interpretation complex (Robinson, 2001).

Against this background, the aim of this chapter is to assess the ^{13}C depletion of soil organic matter under different land cover types. This will help infer a shift in vegetation and serve to trace the source of organic matter from the native vegetation to current transformed vegetation condition. The study also aimed to assess the rate of replacement of organic matter and to establish C partitioning between soil pools of different mean residence times. The ^{15}N depletion in soil and plants was also assessed to gain more understanding of sequestration processes in a plant–soil system, as well as the land-use and cover changes that have occurred over the years in the study area.

5.2 Study area

This study was also carried out at Amakhala reserve from three land cover types which are intact thicket, degraded thicket and grassland.

5.3 Methodology

5.3.1 Soil and foliar sample collection and their preparation

Soil samples were collected in the winter month of August 2011 from the three land covers at depths of 0 – 10, 10 – 20 and 20 – 30 cm. Soils were oven dried at 40 ^{o}C then ground to pass through a 2 mm sieve before analysis for isotope depletion of soil organic matter.

In the case of foliar samples, leaves from three replicate samples of each dominant plant species were collected at each site. These were oven dried at 60 ^{o} C for 48 hours before grinding them to a fine powder in a mill with a 40 µm mesh screen. It was not possible to sample more plant parts as destructive sampling was not permitted in the reserve. All plant and soil samples were then sent to the Council for Scientific and Industrial Research (CSIR)'s Stable Isotope Facility in Pretoria, South Africa for analysis of the natural abundance of carbon and nitrogen isotopes (i.e. $\delta^{13}C$ and $\delta^{15}N$).

5.3.2 $\delta^{13}C$ and $\delta^{15}N$ isotope analysis

Soil samples were treated with HCl first to remove carbonates and then dried before analysis (Boutton *et al.*, 1998). The total C, total N, $\delta^{13}C$ and $\delta^{15}N$ of soil and foliar samples were analysed with an isotope ratio mass spectrometer coupled with an elemental analyser. Data for $\delta^{15}N$ was reported relative to the standard atmospheric N_2 (0 ‰) using the formulae:

$$\delta^{15}N_{sample} = [(R_{sample} / R_{standard}) - 1] \times 1000$$

R = ^{15}N / ^{14}N for the standard or sample.

The results for $\delta\ ^{13}C$ analysis were calibrated against the international standard Vienna Pee Dee Belemnite (V-PBN) as follows:

$$\delta\ ^{13}C_{sample} = [(R_{sample} / R_{standard}) - 1] \times 1000$$

Where R = ^{13}C / ^{12}C for the standard or sample.

*The V-PBN ($\delta\ ^{13}C$) = 0 (Peri *et al*., 2012)

5.3.3 Statistical analysis

Data were analysed for differences between treatment means at $\alpha = 0.05$ significance level. The effects of vegetation transformation and sampling depth on soil and plant isotope ratios were tested and compared using a two-way analysis of variance (ANOVA) test. The potential relationships between $\delta\ ^{13}C$, $\delta\ ^{15}N$, total C, total N and C: N ratios were also examined through correlation analysis using Pearson's Product moment correlation coefficient (r value; $\alpha = 0.05$). All tests were performed with GenStat 14.1 for Windows software.

5.4 Results

5.4.1 ^{13}C isotope depletion of soil organic matter at study sites

Significant differences were recorded in soil ^{13}C isotope depletion at all three sites and depths ($p < 0.001$); with the intact thicket (IT) having more negative ratios at all depths than the other two sites (Figure 5.1 and Appendix H). The isotope depletion increased (became less negative) with increasing depth at the grassland (G) and degraded thicket (DT) but did not change in intact thicket soils. C isotope ratios of IT soils were lowest (more negative) and ranged from -22.98 to -20.96 ‰; these were indicative of mostly C_3 photosynthesis but with some occasional switches to CAM metabolism. DT soils had the highest $\delta\ ^{13}C$ values (least negative) ranging from -20.55 to -12.64 ‰ which would be more characteristic of CAM photosynthesis. C isotope ratios in G soils ranged from -18.5 to -13.78 ‰ and were inferred to have been inherited from predominantly C_4 plants with a few CAM plants in the mix. $\delta\ ^{13}C$ decreased as total C increased in the soil ($r = - 0.67$ at $p < 0.05$ from correlation analysis).

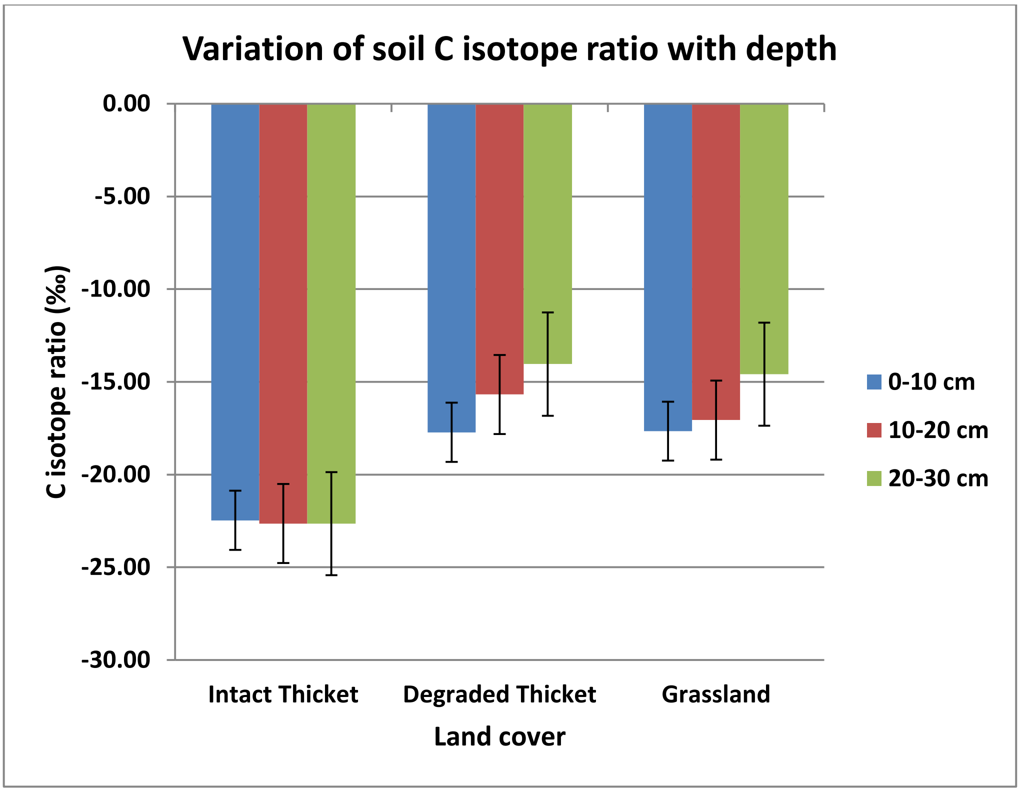


Figure 5.1: Variations in soil C isotope ratio with depth

5.4.2 ^{15}N isotope depletion of soil organic matter at study sites

An opposite trend was observed with soil N isotope ratios between cover types. There were significant differences of soil N depletion across land covers ($p < 0.001$); with IT recording the highest values (more positive ratios) while the DT had the least (less positive) δ^{15}N values at all depths (Figure 5.2 and Appendix I). N isotope ratio also increased with depth at IT and DT ($p = 0.005$). Thus ratios generally increased as the depth increased. The relationship between $\delta\, ^{15}$N and total soil N was weak ($r = + 0.13$; Table 5.3). Soil C: N ratios were highest under IT and lowest under G ($p < 0.001$); and they decreased with increasing depth at all sites ($p < 0.001$).

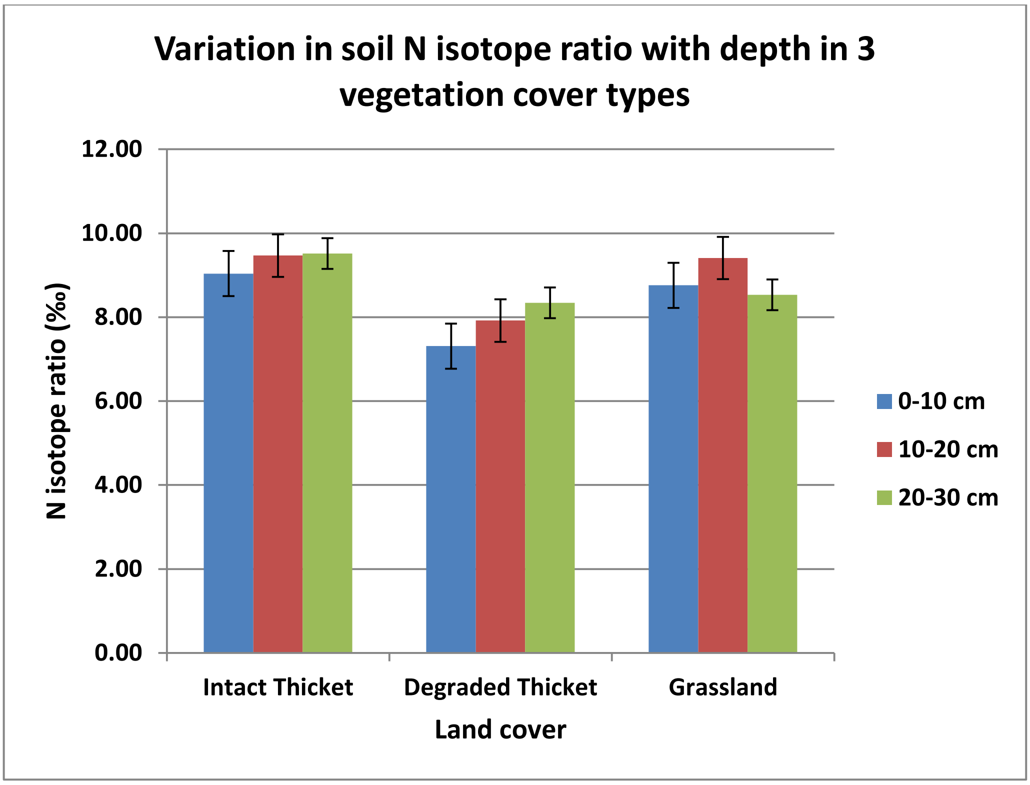


Figure 5.2: Variations in soil N isotope ratio with depth

5.4.3 Plant δ ^{13}C, δ ^{15}N, total C, total N and C: N variability

Plant ^{13}C depletion ratios of dominant plant species showed wide variations between the cover types (p = 0.002, Appendix J). A closer look at the IT showed that all plant guilds did not differ much in δ ^{13}C; and they had an average value of -24.78 ‰ (Table 5.1). Most of the IT plants actually ranged from -27.8 to -23.02 ‰ δ ^{13}C which is indicative of C_3 photosynthesis. It was only the aloe species that had an odd value of -16.34 ‰ that was leaning towards CAM photosynthesis. DT plants however had wider δ ^{13}C ranges of -30.65 to -13.31 ‰. They showed greater diversity in photosynthetic pathways; with the majority of the plants exhibiting more of C_3 photosynthesis due to the more negative ratios. A few IT grass species e.g. *Themeda trianda* and *Panicum maxima* had higher (less negative) δ ^{13}C values which are characteristic of C_4 photosynthesis. The G species all showed δ ^{13}C values leaning towards C_4 photosynthesis (-16.45 to -14.25 ‰).

There was no significant difference in plant δ ^{15}N between the covers (p = 0.36, Appendix K). However, total plant C and N were higher under IT compared to the other 2 covers (Tables 5.1 and 5.2). Foliar C: N ratios were highest in DT and lowest at IT vegetation (p = 0.034).

The plant ^{15}N depletion recorded was positive in all three covers. Generally, δ ^{15}N values increased with increasing plant N content (r = + 0.72); but decreased with increasing C: N ratio (r = -0.64) (Table 5.4). The relationship between δ ^{13}C and plant C showed δ ^{13}C to generally decrease as plant C increased (r = - 0.31). There was a weak relationship between δ ^{13}C and δ ^{15}N in the plants (r = - 0.1).

Table 5.1: Variations in plant δ ^{13}C, δ ^{15}N, total C and total N in intact thicket

Plant Guild	Species	δ ^{13}C (‰)	δ ^{15}N (‰)	Total C (%)	Total N (%)	C: N ratio
Canopy trees	*Euclea undulata*	-27.72	+5.02	53.88	1.29	48.88
	Schotia afra	-27.80	+4.48	51.85	0.99	60.99
	Sideroxylon inerme	-27.04	+6.51	54.08	1.36	46.56
	Guild average	**-27.52**	**+5.34**	**53.27**	**1.21**	**52.14**
Woody shrubs	*Azima tetracantha*	-25.54	+8.00	51.71	1.69	35.78
	Gymnosporia polyacantha	-25.27	+1.60	50.02	0.67	86.51
	Rhus longispina	-24.78	+4.15	51.54	0.72	83.74
	Guild average	**-25.19**	**+4.58**	**51.09**	**1.03**	**68.68**
Succulent shrubs	*Portulacaria afra*	-23.02	+6.37	52.37	1.18	51.81
	Carissa haematocarpa	-25.10	+4.39	53.15	0.60	106.38
	Guild average	**-24.06**	**+5.38**	**52.76**	**0.89**	**79.1**
Ephemerals	Asparagus species	-25.20	+3.40	53.14	0.96	64.36
	Aloe species	-16.34	+6.84	56.08	0.71	92.60
	Guild average	**-20.77**	**+5.12**	**54.61**	**0.83**	**78.48**
Cover range		**-27.8 to -16.34**	**+1.6 to +8**	**50 -56.1**	**0.67 - 1.69**	**35.78-106.38**
Cover average		**-24.78**	**+5.08**	**52.78**	**1.02**	**67.76**

5.4.4 Assessing isotope depletion interactions in plants and soils

A closer look at the correlation analysis showed the following trends in soils. Soil δ ^{13}C decreased (became more negative) as δ ^{15}N increased (became more positive); (r = -0.55) (Table 5.3). It also decreased with increasing total C, total N and the soil C: N (r = -0.64, -0.61 and -0.66 respectively). Soil δ ^{15}N had weak relations with total N and total C; and there was no relationship with C: N ratio in the soil (r = +0.13, +0.14 and 0.05, respectively). Total soil N and C had a highly significant relationship as N increased with increasing soil C (r = 0.99).

In the case of the foliar samples; there was no significant relationship between plant δ ^{13}C and δ ^{15}N (r = -0.1); (Table 5.4). The δ ^{15}N – total N relationship was much stronger in plants than it was in soil; since δ ^{15}N significantly increased with increasing total N (r = +0.72). δ ^{15}N also increased with decreasing plant C: N ratio (r = -0.64). Plant δ ^{13}C decreased as total plant C increased (r = -0.31) as was the case in soil; and also decreased with increasing total plant N (r = -0.21). C: N ratio increased with decreasing plant N (r = -0.9) and slightly increased with increasing plant C (r = +0.13).

Table 5.2: Variations in plant δ ^{13}C, δ ^{15}N, C, N and C: N in DT and G vegetation

Biome	**Species**	**δ ^{13}C (‰)**	**δ ^{15}N(‰)**	**Total C (%)**	**Total N (%)**	**C: N ratio**
Degraded Thicket	*Themeda trianda*	-13.31	+2.27	37.75	0.42	106.16
	Panicum maxima	-13.34	+2.91	48.02	0.50	113.34
	Chrysocoma ciliata	-30.65	+3.59	53.17	0.49	127
	Aloe species	-25.75	+8.54	43.95	1.55	34.02
	Crassula mesembryanthoides	-26.18	+4.88	50.26	0.69	85.03
	Rubus species	-28.21	+5.10	37.12	0.77	55.94
	Senecio species	-30.01	+3.46	50.84	0.87	68.15
	Pteronia incana	-28.94	+2.76	52.19	0.48	127.57
	Mesembryanthemacae species	-26.99	+6.52	55.35	0.52	123.94
	Metalasia species	-27.83	+0.81	52.07	0.43	140.25
	Cover range **Cover average**	**-30.65 to - 13.31** **-25.65**	**+0.81 to +8.54** **+3.98**	**37.12 to 55.35** **48.49**	**0.42 to 1.55** **0.64**	**34.02 to 127.57** **103.7**
Grassland	*Cyanodon dactylon*	-14.74	+4.74	46.40	0.56	97.01
	Fabaceae species	-16.45	+4.61	50.36	1.08	54.19
	Panicum maxima	-14.34	+3.0	48.02	0.52	92.35
	Atriplex pecies	-14.25	+2.9	49.0	0.50	98
	Cover range **Cover average**	**-16.45 to -14.25** **-14.94**	**+2.9 to +4.74** **+3.81**	**46.36 to 50.36** **48.45**	**0.50 to 1.08** **0.67**	**54.19 to 97.01** **85.39**

Table 5.3 Soil $\delta\,^{13}$C, $\delta\,^{15}$N, total C and total N correlations

	Correlation coefficient (r) at $p < 0.05$				
$\delta\,^{13}$C	-				
$\delta\,^{15}$N	-0.55***	-			
%C	-0.64***	+0.14*	-		
%N	-0.61***	+0.13*	+0.99***	-	
C: N ratio	-0.66***	0.05^{ns}	+0.80***	+0.74***	-
	$\delta\,^{13}$C	**$\delta\,^{15}$N**	**%C**	**%N**	**C: N ratio**

*** Strong, ** medium, *small, ^{ns} no correlation

An overview of the plant - soil trends gave the summary of more positive $\delta\,^{15}$N values in IT, list positive values in DT and moderate $\delta\,^{15}$N in G for both soils and plants (Table 5.5). The trend was somewhat slightly variable for $\delta\,^{13}$C. The $\delta\,^{13}$C was consistent at IT giving averages of -22.58 and -24.78 ‰ in soil and plant respectively. G also gave consistently higher $\delta\,^{13}$C of -16.44 ‰ in soil and -14.94 ‰ in plants. DT however that had the least negative $\delta\,^{13}$C average of -15.81 ‰ in soil; had a lower plant $\delta\,^{13}$C average of -25.65 ‰.

Table 5.4 Plant $\delta\,^{13}$C, $\delta\,^{15}$N, total C and total N correlations

	Correlation coefficient (r) at $p < 0.05$				
$\delta\,^{13}$C	-				
$\delta\,^{15}$N	-0.1*	-			
%C	-0.31**	$+0.099^{ns}$	-		
%N	-0.21*	+0.72***	+0.11*	-	
C: N ratio	$+0.04^{ns}$	-0.64***	+0.13*	-0.90***	-
	$\delta\,^{13}$C	**$\delta\,^{15}$N**	**%C**	**%N**	**C: N ratio**

*** Strong, ** medium, *small, ^{ns} no correlation

Thus, there was a very significant decrease (more negative values) in $\delta\,^{13}$C as one moved from the plants to soil in DT giving an enrichment ratio (ER) of -9.84 ‰ (Table 5.5). This difference was much smaller in IT, while the G site did not show much $\delta\,^{13}$C variation between plants and soil. In the case of $\delta\,^{15}$N; values were reduced as one moved from the plants to soil giving negative ERs across all sites.

Table 5.5 Summary of isotope depletion ratios for plants and soil

Component	Isotope	Intact thicket	Degraded thicket	Grassland
Soil	$\delta\,^{13}C$ (‰)	-22.58	-15.81	-16.44
	$\delta\,^{15}N$ (‰)	+9.35	+7.86	+8.90
Plants	$\delta\,^{13}C$ (‰)	-24.78	-25.65	-14.94
	$\delta\,^{15}N$ (‰)	+5.08	+3.98	+3.81
Isotope ERs	$\delta\,^{13}C$ (‰)	-2.2	-9.84	+1.5
	$\delta\,^{15}N$ (‰)	-4.27	-3.88	-5.09

*Isotope enrichment ratio (ER) = isotope ratio in plant – isotope ratio in soil

5.5 Discussion

The soil C isotope ratios indicated that the original vegetation in the IT had intermediate photosynthesis that switched between C_3 and CAM photosynthesis. Plant values however showed a shift towards more of the C_3 photosynthesis since $\delta\,^{13}C$ values were becoming more negative in plants. This shift was even higher at the DT site. DT soils exhibited the highest $\delta\,^{13}C$ signatures in initial vegetation with an average soil $\delta\,^{13}C$ of -15.81 ‰; which typically represented a predominance of plants with CAM metabolism. The higher soil $\delta\,^{13}C$ at DT actually indicated that this site used to have proportionally more CAM plants than IT. However, the current DT vegetation has more negative C isotope ratios with some reaching levels as low as -30 ‰. This shows a substantial increase in the proportion of C_3 plants at this site. The G on the other hand showed little variation between the original and current vegetation as the C signatures recorded C_4 metabolism in both plants and soils. A shift to predominantly C_3 metabolism is not desirable as C_3 plants are not as water use efficient as their CAM counterparts. Mooney *et al*. (1977) explained that CAM plants in arid regimes fixed large amounts of CO_2 at night as an adaptive mechanism to conserve water. The assimilation of carbon dioxide at night by CAMs when potential evapotranspiration from the plant is lowest ensures that the amount of water lost per unit of carbon gained is relatively low. C_3 plants on the other hand do not possess this trait, so they can only photosynthesize during the day when evaporation rates are too high; making them less water-use efficient and poorer at adapting to prolonged dry periods. As a result, in the event of recurrent droughts, C_3 plants will struggle to cope under limited water supply since they are not as water-use efficient as CAM plants.

The divide between C_3 and CAM photosynthesis is not rigid though. Farquhar *et al*, (1989) explained that upon exposure to increased drought, some leaf succulents of arid habitats could shift from C_3 to CAM photosynthesis with this shift being reversible, depending primarily on plant water status. They further explained that other plants exhibited CAM metabolism as epiphytic juveniles but later switched to C_3 metabolism when roots were more developed. Because of this, what might appear as C_3 plants from the $\delta\ ^{13}C$ ranges could be sometimes facultative CAM metabolism since such plants are known to easily switch between C_3 and CAM depending on environmental conditions.

According to Griffiths (1992); C_3 plants have $\delta\ ^{13}C$ values of -32 to -22‰ (mean -27‰) , C_4 plants range from -17 to -9‰ (mean -13‰); obligate CAM plants have $\delta\ ^{13}C$ values similar to C_4 plants; while facultative CAMs may range from -30 to -10‰, depending on the relative amount of carbon fixed by CAM versus C_3 photosynthesis. In their study of South Africa Fish River thicket, Mooney *et al.* (1977) found some woody shrubs that were predominantly C_3 (ranges of -26.4 to -23.6 ‰ $\delta\ ^{13}C$); while the majority of the succulents were intermediate between C_3 and CAM (-19 to -15‰ $\delta\ ^{13}C$) and others were predominantly CAM (-14 to -13 ‰ $\delta\ ^{13}C$). They concluded that not all succulent plants had CAM metabolism. Mooney *et al.* (1977) further discovered that the smaller, leafier succulents exhibited greater flexibility between C_3 and CAM photosynthesis while the more massive forms showed least carbon isotope fractionation indicative of predominantly CAM metabolism. They explained that the flexible metabolism by smaller leafy succulents was determined by their internal water supply; thus they have predominantly daytime (C_3) photosynthesis when the water supply is abundant but can shift to night-time carbon fixation (CAM) when water becomes limited.

Studies show that some succulent plants including *Portulacaria afra* have metabolism intermediate between C_3 and CAM (i.e. facultative CAM); some are obligate CAM e.g. *Crassula ovata*, some such as *Euclea undulate*, *Azima tetracantha* and *Rhus longispina* are obligate C_3 plants; while others e.g. *Themeda trianda*, *Panicum maxima*, *Cyanodon dactylon* and Atriplex species are C_4 plants (Palmer, 2004; Mills *et al.*, 2005). The present study confirmed most of these findings. However, *Portulacaria afra* showed slight deviations from that reported in literature since it had a more negative $\delta\ ^{13}C$ value of -23 ‰, which is indicative of mostly C_3 photosynthesis. Troughton *et al.* (1977) explained that reproductive tissue was more likely to have a less negative ^{13}C value than leafy tissue of the same plant. Since we only looked at the leaf tissue, this could explain the more negative values we got for some of the species that are known to exhibit CAM metabolism. Marais *et al.* (2009)

explained that *Portulacaria afra* mostly used C_3 photosynthesis when soils were wet; making it more efficient than plants that only used CAM metabolism (i.e. the obligate CAM plants).

What came out clearly in our study is the shift in isotopic composition between plants and soils. From the isotope enrichment ratios, it is evident that major shifts in vegetation composition and metabolism have occurred especially at the DT site. Enrichment ratios recorded were highest at DT (-9.84‰) and lowest at G (+1.5‰) showing greater changes in vegetation at DT. Boutton *et al.* (1998) also found long-term heavy grazing by livestock to increase the proportion of C_3 shrubs in an area that was previously dominated by C_4 grasses. In the present study, heavy browsing by wildlife at DT may be responsible for altering predominantly CAM vegetation and replacing it with C_3 shrubs and herbs. The disappearance of succulent CAM plants from the arid ecosystem is not desirable as these can proliferate better under limited moisture conditions in comparison with C_3 species as already mentioned before. Moreover, major shifts in native vegetation interfere with the stabilisation of carbon stock due to a reduced sequestration potential (Lechmere-Oertel *et al.* 2005). Given a scenario of declining rainfall that was shown with the overall pooled data for the different rainfall stations (chapter 3), thicket species from transformed and degraded land would really struggle to recover to their original state due to the limiting conditions of moisture and declining fertility under degradation. Dramatic changes in ecosystem structure also have the potential to negatively influence the hydrology, biodiversity, landscape evolution and future land use options of the affected areas in the long term as the overall productive potential of the land and water availability would have been diminished (Ehleringer *et al.* (1998).

Soil $\delta^{13}C$ generally increased with increasing depth, more so under DT. This increase further confirmed that the deeper soil layers with higher $\delta^{13}C$ at DT were derived from CAM dominated vegetation while the top soil layers showed less of CAM signalling a shift to current C_3 vegetation. There is wide evidence of increasing ^{13}C abundance at greater soil depths or along a continuum of more decomposed SOM fractions. Boutton *et al.*, (1998) stated that recent changes in relative C_3: C_4: CAM productivity might only be evident in the $\delta^{13}C$ of organic C near the soil surface; where OM turnover is most rapid and new organic matter inputs are concentrated. OM turnover in the top layer was thus high due to inputs from current vegetation. On the other hand; isotopic signatures from preceding plant communities may persist deeper in the soil profile, making the OM turnover rates at deeper layers lower. Deeper soil layers thus had a record of older OM residues contributed from previous CAM vegetation. Michener and Lajtha (2007) postulated several mechanisms for ^{13}C enrichment at greater depth such as soil ^{13}C enrichment as a result of fractionation during OM decomposition, mixing of new C inputs with older OM that has a different δ ^{13}C, and preferential decomposition of SOM by soil microbes. The negative correlation between soil δ ^{13}C with both total C and C: N ratio, further confirms that C_3 wood C instead of the original CAM carbon was mainly responsible for the current soil C contributions.

Results from ^{15}N depletion were useful in explaining the N mineralization processes occurring in the study area. Soil δ ^{15}N was significantly higher at IT and lowest at DT ($p< 0.001$). This corresponded with higher soil N in IT. Michener and Lajtha, (2007) reported soil ^{15}N values to be generally lower in areas with low N values and closed N cycles than in relatively N-rich environments with more open N cycles. Thus the ^{15}N depletion will give an idea of the rate of mineralization occurring in an area, with more positive values showing higher N mineralization which was the case in IT. Kahmen *et al.* (2008) also observed increasing soil δ^{15}N with increasing rates of soil N cycling. They explained that high soil N mineralization rates often correlated with denitrification or leaching of inorganic N; leading to losses of ^{15}N-depleted N, leaving the remaining inorganic N pool enriched in ^{15}N. Plant ^{15}N depletion did not show any significant differences between the sites however.

The results showed that plants generally had lower δ ^{15}N values than soils (represented by the negative ER values). Pardo *et al.* (2007) explained that this was because plants assimilate inorganic N, which as a result of microbial transformations is depleted in ^{15}N relative to soil. They further stated that microbes discriminate against the heavier ^{15}N during microbial transformations of N in the soil. Thus plant δ ^{15}N values typically reflect the δ ^{15}N of the inorganic N they take up. As a result, the trend for foliar δ ^{15}N was similar to that of soil with the IT having the most positive δ ^{15}N values, while DT had lower values. This is because litter inputs are a strong determinant of the δ ^{15}N of surface soil horizons; hence ^{15}N-depleted litter will lower the δ ^{15}N of the organic soil horizons, while ^{15}N-enriched litter will lead to ^{15}N enriched surface horizons (Pardo *et al.*, 2007). Cloern *et al.* (2002) also found the smallest δ ^{15}N values in riparian vegetation which was in soil depleted in ^{15}N relative to soil N; while high δ ^{15}N was measured in plants that assimilated inorganic N from aquatic sediments where microbial mineralization, nitrification and denitrification produce ^{15}N enriched inorganic N.

Natural ^{15}N abundance in soils and plants of terrestrial systems is determined by long-term changes in rates of N inputs, transformations, losses and is an indicator of N source. It is regulated by internal N cycling processes such as mineralization, nitrification, leaching, gaseous losses and inputs as well as by extrinsic factors such as fertilization, climate, soil age and grazing (Xu *et al*., 2010). Herbivores play an important role in regulating N cycling in rangelands through grazing, trampling (both of which alter plant species composition) and faecal droppings, all of which ultimately affect plant litter decomposition or soil microbial activities (Xu *et al*., 2010). In the present study, the DT site was degraded by heavy browsing which could have resulted in reduced ^{15}N enrichment in its soils. Frank *et al.* (2000) also found positive correlations of soil ^{15}N levels with net N mineralization and net nitrification rates in ungrazed but not in grazed grasslands. Xu *et al*. (2010) noted high-intensity grazing by sheep to result in a significant decrease in δ^{15}N of surface soils in grassland. They attributed the interaction between aridity and heavy grazing to have played an important role in reducing δ^{15}N of surface soils of semi-arid grassland.

The close relation between foliar δ^{15}N and soil δ^{15}N suggests that N was efficiently recycled from SOM; more so at the IT since it had higher δ^{15}N values in both plants and soils. An efficient cycling of N implies that the soil C and N cycles are closely linked (Peri *et al*., 2012). The high soil δ^{15}N at IT reflects increasing soil N cycling which is associated with losses of ^{15}N-depleted mineral N that lead to a gradual δ^{15}N enrichment of the remaining soil N (Kahmen *et al*., 2008), and represents a more open N cycle at IT than at the DT and G. Higher N cycling imply higher N availability leading to improved plant biomass accumulation and higher C stabilisation of the ecosystem.

There was also enrichment of ^{15}N with increasing soil depth. This only served to confirm that the composition of original vegetation at the sites (represented by more positive δ^{15}N at greater depth) is different from the current vegetation (with lower δ^{15}N values at the surface horizon); showing vegetation shifts over the years. Brenner *et al*. (1999) also found the highest δ^{15}N at greater depth; with values steadily declining above this depth. The more negative ERs from plant-soil comparisons also show shifts in ^{15}N source between original and current vegetation.

The three sites were significantly different in their foliar total N ($p = 0.036$) and C: N ratio ($p = 0.034$) but did not differ in the foliar C and δ^{15}N ($p > 0.05$). Foliar C: N ratio was inversely related to total N ($r = -0.9$). Thus greater soil N availability at IT resulted in leaf litter with higher N concentrations and lower foliar C: N ratios. Michener and Lajtha, (2007) explained that greater N availability in a soil could contribute to the partitioning of soil C to SOM resulting in greater stabilization of the C stock in a soil. This could explain the higher soil C at IT site. Nitrogen availability indirectly controls soil C dynamics through its effects on organic matter decomposition; thus higher litter N gives lower C: N ratios that accelerate initial stages of litter decomposition making N become readily available for plant uptake resulting in accelerated plant growth (Khalila *et al.*, 2005). High C: N ratios on the other hand immobilise N in organic residues such that it becomes unavailable for mineralization and is not readily available for plant uptake. This gave relatively lower N values in DT soils (Chapter 4) with correspondingly higher plant C: N ratios. Optimum residue C: N ratios for efficient microbial decomposition are in the ranges of 30-35:1, with a higher ratio causing slower decomposition rates.

5.6 Conclusion

$\delta\ ^{13}C$ values of soil organic matter and plant foliar in conjunction with their $\delta\ ^{15}N$ were used to infer the vegetation changes that have occurred over the years at the sites. The results obtained clearly showed that soils at the IT and DT retained a strong isotopic memory of the CAM vegetation that once occupied this area; while soils at the grassland site showed a predominance of C_4 plants as the original vegetation. The conclusion drawn from this study was that organic matter inputs from the current vegetation at the IT and DT are not in isotopic equilibrium with soil organic carbon. It could also be possible that there was preferential decomposition of soil organic matter by microbes leading to ^{13}C enrichment at deeper soil depths. Whichever is the case, it was evident that land that used to be dominated by CAM plants is now occupied by plants that have shifted more towards C_3 metabolism. Variations (increases) of both $\delta\ ^{13}C$ and $\delta\ ^{15}N$ with soil depth and decreases in their respective enrichment ratios provided further evidence that the sources of these isotopes have changed over the years. This shift was more substantial at the DT as indicated by the more negative plant $\delta\ ^{13}C$ and overall ERs. Thus, species composition, abundance and metabolism have been drastically altered at DT through degradation by wildlife herbivores. A shift from CAM to predominantly C_3 metabolism is not desirable since C_3 plants are not as water-use efficient as CAM plants. As a result their resilience to the drier conditions typical of the study area is limited. This reduces their net primary productivity and propagation potential. It was observed from the previous chapters that vegetation transformation is prominent in the area and that rainfall at the reserve is quite low with a general declining trend over the broader

Albany region. In order to avoid further deterioration of the land, more effort needs to be directed towards preserving the vegetation in its original form since this ensures better water-use efficiency by the plant. Preservation ensures that the succulent and canopy species that are better at tolerating marginal conditions of moisture and fertility are retained, thereby curbing further climate shifts. The G site proved to have remained stable over the years since it retained a record of C_4 plants in its soils, which is similar to the current vegetation. However, its poor fertility status and sandy soils resulted in reduced C accumulation in its plants and soils (chapter 4). The natural abundance of the stable ^{15}N isotope in soils and plants emerged as a simple tool to assess ecosystem N dynamics; with higher $\delta\ ^{15}N$ values showing greater N mineralization at the IT site. Herbivores through their grazing, trampling and faecal matter altered plant species composition ultimately reducing N cycling and C stabilisation at the DT.

CHAPTER 6: THE IMPACT OF LAND-USE TYPE AND SOIL PROPERTIES ON CO_2 EFFLUXES FROM THE SOIL

The previous chapter gave detailed information on the photosynthetic pathways of the dominant species in the different land cover types and how they compare with the isotopy of their soil organic matter; in an attempt to study the land-use history of the study area. This chapter now addresses the third objective which is to investigate the impact of these land-use / cover types and soil properties on CO_2 effluxes from the soil. It was necessary to see if the different land covers would have a bearing on the amount of CO_2 emitted from the soil. The objective was achieved by measuring the CO_2 evolved from soil of different land covers namely intact thicket, degraded thicket and grassland. The dynamic chamber method was used in the assessment. Particular attention was given to the influence of soil properties (i.e. chemical, physical and thermal properties) and vegetation type on CO_2 effluxes. All this was done to see the optimum combination of properties and conditions that led to the least amount of CO_2 emissions from the soil.

6.1 Introduction

The atmospheric concentration of CO_2 among other greenhouse gases (GHGs) has risen substantially over the years. Tripati *et al*. (2009) stated that atmospheric CO_2 reached a concentration of 387 ppm in 2009, which was much higher than its natural pre-industrial level by more than 105 ppm. Another study by Peters *et al*. (2012) revealed that global carbon dioxide from fossil fuel combustion, cement production, deforestation and land use emissions have increased to 389.6 ppm in 2010. They explained that fossil fuel emissions had risen by 3.1 per cent each year between 2000 and 2010, a figure that is three times the rate of increase during the 1990s. Increase in GHG emissions through anthropogenic activities is not desirable as it is often associated with rising air temperatures and causes major disturbances to precipitation patterns. Allison *et al*. (2009) reported that global temperatures have increased at a rate of 0.19°C per decade over the past 25 years, a trend that is proportional to the increase in anthropogenic GHG emissions over the years. Allan and Soden (2008) observed a distinct link between extreme precipitation events and human induced warming through satellite observations and climate model simulations, with heavy precipitation events increasing during warm periods and decreasing during cold periods.

Historical records demonstrate that inter-annual rainfall variability over Zimbabwe, South Africa and other parts of southern Africa is increasing, resulting in flood years and droughts becoming more frequent and severe. Mason and Jury, (1997) observed increased drying over Zimbabwe, northern Botswana and eastern South Africa since the late 1970s. There is further evidence of changes in seasonality and weather extremes (Washington and Preston, 2006). Cai *et al*. (2012) reported that since the late 1970s, the semi-arid regions of southern Africa have experienced a drying trend in austral autumn, particularly in April and May. Morishima and Akasaka (2010) also showed the spatial and temporal seasonal trends of rainfall in Angola, Zambia and Namibia tending to decrease from December to March and surface temperature from Namibia to south-eastern South Africa tending to increase from July to October.

Soils play a crucial role as sources and sinks of CO_2. Yuste *et al.*, (2007) described soil as the largest terrestrial C pool, which is vulnerable to climatic warming making its potential loss amplify further warming. It is thus essential to assess the amount of CO_2 emitted from soil and to what extent soils serve as a sink for GHGs. The major sources of CO_2 in the soil include respiration of plants roots, soil fauna and microorganisms. Sulzman *et al.*, (2005) found that the contribution of root respiration, above and below-ground litter decomposition to total soil respiration was 23, 19 and 58% respectively. Several soil properties affect soil respiration and these include temperature, moisture, pH as well as organic matter content and composition (Hillel, 2003). Both temperature and moisture usually increases respiration since they have a positive effect on organic matter mineralization and microbial activity (Schaefer *et al.*, 2009). On the other hand, soil respiration increases with increasing soil moisture up to a level where pores become filled with too much water that limits oxygen availability and interferes with the organism's ability to respire. Hence in dry soils, respiration declines because the soil moisture deficit limits microbial activity; while the low oxygen levels of extremely wet soils result in poor organic matter decomposition (Yuste *et al.*, 2007). Nkongolo (2010) went on further to assess the effect of soil thermal properties such as diffusivity, conductivity and resistivity and found them to be significantly correlated with soil CO_2 effluxes when temperature could not account for this variation. Gaseous transport processes in the soil have also been known to be affected by total porosity, pore size distribution, and tortuosity of continuous air-filled pore space (Nkongolo *et al*, 2010; Hillel, 2003). The pore tortuosity factor in particular defines the pathway of gas diffusion, and is affected by pore geometry; thereby giving rise to differences in diffusion rates between poor and well-structured soil (Bartjes and Bridges, 1992).

Management practices such as cultivation, land-use change and residue incorporation all affect soil respiration rates as they have an impact on the organic matter status of the soil. Litter fall for example affects soil organic matter content, pH, temperature and moisture; thus soil respiration varies remarkably with change in vegetation cover. Schaefer *et al.,* (2009) found above-ground plant litter removal to result in a marked reduction in soil respiration. Land management practices such as tillage, residue burning and removal diminish SOM content and microbial activity by reducing aggregate stability and porosity consequently reducing respiration (USDA, 2009). There is evidence of different respiration rates among different vegetation cover or land-use types. Sundarapandian and Kirthiga (2011) explained that different land use types will differ in vegetation structure and species composition, density of above and below ground biomass, the amount and quality of organic substrate available to soil microbes, as well as physico- chemical properties of the soil; all of which ultimately affect net CO_2 exchange. It is against this background that it was necessary to assess soil respiration rates in different vegetation environments.

The aim of this present study was therefore to assess the effect of soil properties and vegetation cover type on CO_2 effluxes from the soil.

6.2 Study area

To ensure easier monitoring, this study was also carried out at Amakhala reserve using the intact thicket (IT), degraded thicket (DT) and grassland (G) as study sites. The reserve was chosen for its logistical convenience, since it was much closer to the University and allowed for the repeated measurements of CO_2 effluxes that were required in this experiment.

6.3 Methodology

6.3.1 Soil CO_2 assays / measurements

Soil CO_2 flux measurements were done at each of the three sites using the dynamic chamber method. Five points arranged in a transect were chosen at each site, giving a total of 15 gas monitoring points. CO_2 effluxes were measured using an Automated Carbon dioxide Exchange (ACE) Analyser. This instrument measures soil respiration and expresses it as net carbon dioxide exchange rate (NCER). The ACE station has an electronic box that comprised a 240 x 64 pixel LCD display, a five key button keypad, a microprocessor control system and an infra-red gas analyser (ADC Bioscientific, 2011). It makes periodic CO_2 measurements (Assays) at pre-determined intervals, and is powered by a car battery. Coupled to the ACE station was a photosythetic active radiation (PAR) sensor for measuring PAR, and a soil temperature thermistor for measuring temperature during each assay (ADC Bioscientific, 2011). The chamber also comprised a 1 Litre stainless steel soil collar that was inserted into the soil prior to measurements, to accurately define the soil area being analysed.

During each assay, the open mode (with a zero function added to measure ambient CO_2 at the soil surface) was used since it gave more accurate results than the closed mode. As such, the rate of change of CO_2 gas concentration in the chamber was measured (ADC Bioscientific, 2011). Assays were done at 20-30 day intervals between 10 am and 1600 hours over a period of 10 months (June 2011 to March 2012). Variables measured in each assay included soil temperature, moisture, NCER (in $\mu molsm^{-2}sec^{-1}$), zero / ambient CO_2 denoted ΔCO_2 (in $\mu molsm^{-3}$), PAR denoted Q (in $\mu molsm^{-2}sec^{-1}$), flow rate into chamber denoted μ (in $\mu molsec^{-1}$) and soil penetration resistance (in Kg cm^{-2}). Soil moisture was measured with an MPM-100 moisture probe meter, and penetration resistance was measured using a penetrometer. Penetration resistance was important since it gave an idea of the degree of compaction of the soil which affects gas diffusivity. CO_2 effluxes for each site were also analysed for seasonal variations and comparisons were made between cover types and measured soil variables.

6.3.2 Statistical analysis

Measurements of NCER, PAR, μ, soil temperature, moisture and penetration resistance at each gas monitoring point were averaged for each land-cover type and sampling date to give a monthly mean. The monthly NCER means were then averaged to give seasonal effluxes. Data was analysed for variability using the Statistica software, Version 11 through an analysis of variance (ANOVA) test; and by determining the degree of correlation between CO_2 flux, soil properties and land-cover type using Pearson's product moment correlation coefficient (r value) (Hill and Lewicki, 2007). A multiple linear regression analysis was also done to determine the impact of the different environmental variables on CO_2 fluxes. All tests were done at $p < 0.05$ significance level. Some 2-D graphs and scatter plots of NCER against each treatment variable of soil temperature, moisture and penetration resistance were also plotted using the Genstat statistical software, with month and cover type being the grouping variables.

6.4 Results

6.4.1 Variation in soil moisture and its effect on NCER

Soil moisture increased in the winter months of June to August (averages of 5 to 16 %), and was then quite low (averaged 2 to 9 %) in spring from September to November at most of the study sites (Figure 6.1). The summer months of December - January had considerably below average moisture (2 to 8 %); particularly at the G and DT. Moisture rose to high levels in February-March with the highest moisture being recorded in March 2012 (average 13 to 16 %), while September was the driest month (average ~ 2 %) at all sites. The overall trend was a general increase in moisture as we moved from June to March as shown by the correlation ($r = +0.26$, $p < 0.05$; Table 6.1). Specific seasonal moisture variations increased in the order autumn > winter > summer > spring. Soil moisture seemed to decrease as one moved from the IT through DT to G ($r = -0.23$, $p < 0.05$). The NCER was positively influenced by the progress of soil moisture, with effluxes rising with increasing soil moisture as shown by the scatter plot (Appendix L) and correlation analysis ($r = +0.35$, $p < 0.05$; Table 6.1). Regression analysis also showed that soil moisture had a strong positive relation with effluxes ($R^2 = 0.83$, $p < 0.01$; Appendix O).

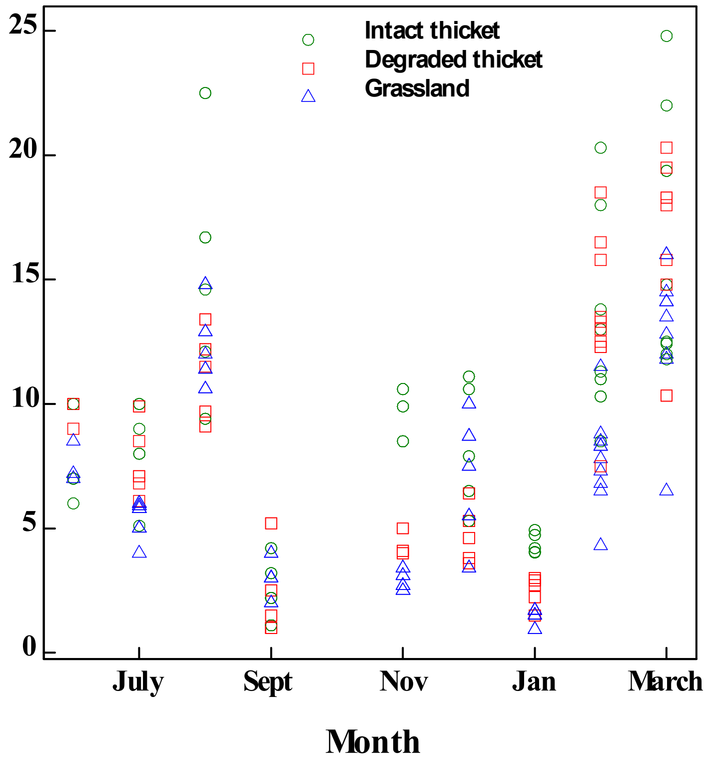


Figure 6.1 Variations of soil moisture with time at three sites

6.4.2 Variation in soil penetration resistance and its effect on NCER

Penetration resistance responded negatively to soil moisture ($r = -0.53$ at $p < 0.01$; Table 6.1), with soil becoming harder as it dried. The highest penetration resistance (~ 14 Kgm^{-2}) was thus recorded in September since this month had the lowest soil moisture in DT; while the lowest penetration resistance (3 to 4 Kg m^{-2}) was recorded in March at both the G and IT as it had the highest soil moisture on record (Figure 6.2). Because of this, NCER tended to decrease with increasing soil penetration resistance ($r = -0.29$ at $p < 0.05$; Table 6.1). Regression analysis also showed a strong effect of penetration resistance on NCER, $R^2 = 0.43$, $p = 0.044$ being recorded (Appendix O). Penetration resistance generally increased as one moved from IT through G to DT ($r = +0.38$; $p < 0.001$), again showing the reverse trend for moisture. The DT recorded the highest penetration resistance for all seasons, while IT had the lowest values which hardly varied between seasons. The general seasonal trend of increased resistance is in the order spring > summer > winter > autumn at the G and DT.

Table 6.1. A Pair-wise comparison of Pearson's product moment correlation analysis (r value) for all measured variables

	Land Cover	Cref	Moisture	Month	NCER	Pen Rest	Q	Temp	μ
Land Cover	-								
Cref	-0.008	-							
Moisture	-0.23*	-0.02	-						
Month	-0.05	-0.54***	+0.26*	-					
NCER	-0.08	-0.22*	+0.35**	+0.31**	-				
Pen Rest	+0.38**	-0.14	-0.53***	-0.05	-0.29*	-			
Q	+0.15	-0.33**	0.08	+0.10	-0.04	+0.16*	-		
Temp	-0.02	-0.60***	0.04	+0.58***	0.08	0.07	+0.53***	-	
μ	-0.1	-0.04	-0.03	0.004	-0.03	-0.04	-0.07	0.007	-
ΔCO_2	-0.09	-0.17*	+0.33**	+0.28*	+0.99***	-0.30**	-0.06	0.05	0.04

*Marked correlations are significant at *** $p<0.001$, ** $p<0.01$ and *$p < 0.05$

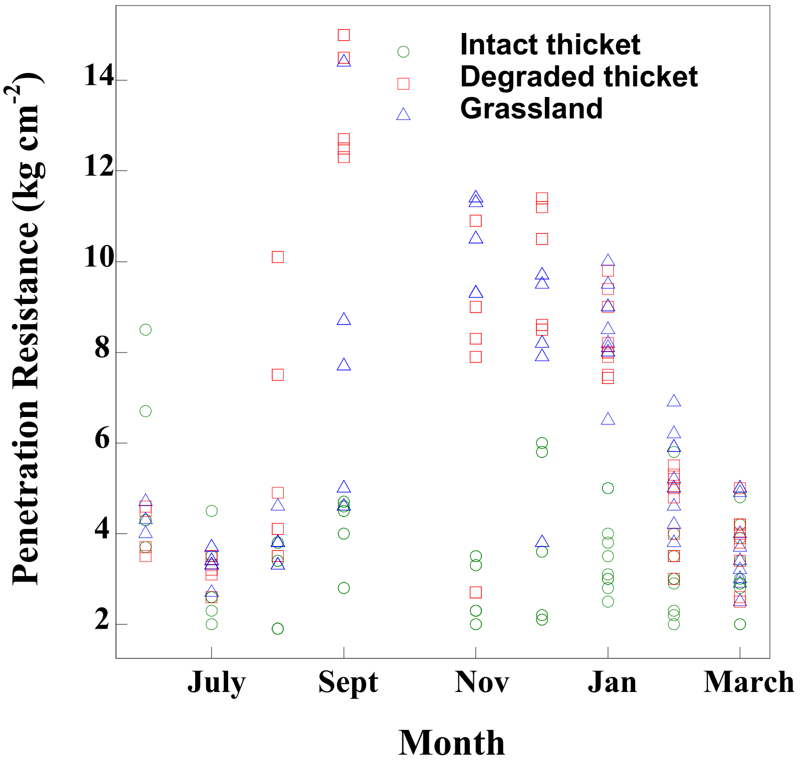


Figure 6.2 Variations of soil penetration resistance with time at three sites

6.4.3 Variation in soil temperature and its effect on NCER

Although temperature increased ($r = +0.58$, $p < 0.001$), (Figures 6.3 and Appendix N) from June to March; it had no significant overall effect on NCER. It was only highly correlated with photosynthetic active radiation, Q ($r = +0.53$, $p < 0.05$; Table 6.1). The influence of temperature only became apparent after a closer look at monthly effluxes. The winter June and July fluxes for example; responded well to rising temperatures with a correlation of above +0.5 in both cases; as so did the February (r = +0.31); November and December (r = +0.13 for both months) effluxes.

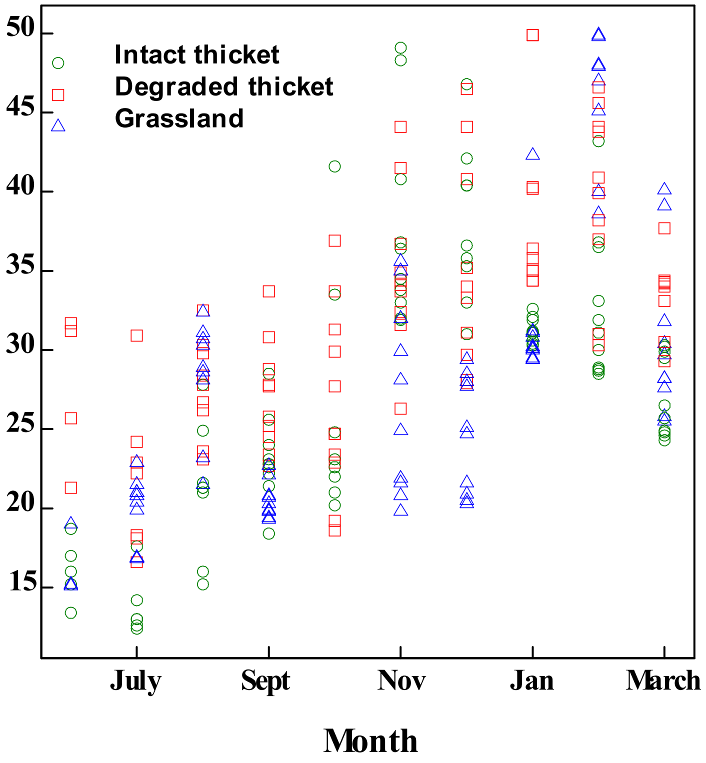


Figure 6.3 Variations of soil temperature with time at 3 sites

6.4.4 Variations in NCER over time

Seasonal variations of NCER were characterized by a general winter (June to August) drop at all three sites (Figure 6.4). Spring effluxes experienced a slight rise in September then a drop in October to November, with the October-November period experiencing very low CO_2 effluxes (average ~ 0.2 µmolsm^{-2}sec^{-1}) at the DT and G. In the case of IT however; the spring rise occurred in October (average of 0.8 µmolsm^{-2}sec^{-1}) while September and November had more or less similar amounts (average ~0.6 µmolsm^{-2}sec^{-1}). In summer there was a rise in NCER in December followed by a drop in January before rising again in February-March at all sites. The highest CO_2 effluxes were recorded during the February-March period. Gas effluxes generally rose as one moved from June to March (r = +0.31, $p < 0.001$; Table 6.1) increasing with moisture (r = +0.35, $p < 0.05$) but decreasing with high penetration resistance (r = -0.29, $p < 0.05$).

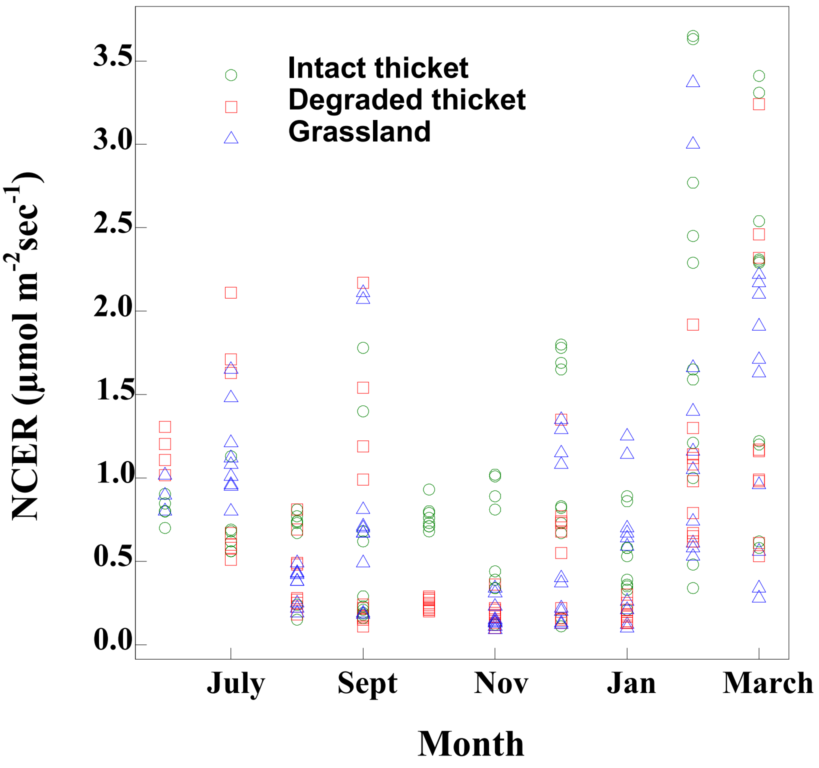


Figure 6.4 Variations of NCER with time at 3 sites

No clear overall variations in NCER were observed between the three vegetation covers (r = -0.08 at $p < 0.05$; Table 6.1); what varied rather were the seasonal trends. The G and DT followed a more or less similar trend i.e. high NCER values in the June-July winter (average 0.9 to 1.2 µmolsm^{-2}sec^{-1}) then declines as low as 0.2 to 0.8 µmolsm^{-2}sec^{-1} from August to January followed by further increases averaging above 1 µmols m^{-2}sec^{-1} in February - March. The IT on the other hand started off with very low NCER values of below 1 µmols m^{-2}sec^{-1}in winter (June-August) and Spring months (September-November); followed by a slight rise to 1 µmols m^{-2}sec^{-1} in December, a slight decrease in January then a sharp increase averaging 1.3 to 2 µmols m^{-2}sec^{-1} in February-March.

The overall seasonal gas effluxes increased in the order autumn > summer > winter > spring emissions (r = +0.7; Tables 6.2 and 6.3). Specific site - seasonal interactions were lower winter effluxes for IT than DT and G; while the reverse was true for summer and autumn effluxes. Spring effluxes were variable however with October - November effluxes being higher in IT than DT and G; while the opposite was true for September effluxes.

6.4.5 Trends of other measured variables

The Cref generally decreased with time ($r = -0.54$ at $p < 0.05$) and most of the other measured variables such as temperature ($r = -0.6$), Q ($r = -0.33$), NCER ($r = -0.22$) and ΔCO_2 ($r = -0.17$), but did not vary between sites (Table 6.1). ΔCO_2 closely followed the trend for NCER with a correlation of almost 1 ($p < 0.001$). Q positively responded to temperature ($r = +0.53$) and increased with time ($r = +0.1$), penetration resistance ($r = +0.16$) and as one moved from IT through DT to G ($r = +0.15$). The flow rate into chamber (μ) was not affected by any of the measured variables (Table 6.1). Multiple regression analysis also showed Cref and ΔCO_2 to be significantly related with NCER variability ($p < 0.001$, Appendix O).

6.4.6 Summary of seasonal variations

As shown in Table 6.3, the months during which assays were done were divided into Winter (June-August), Spring (September-November) and Summer (December-February) seasons. The autumn data was only collected for March owing to logical constraints. NCER values were further converted to units of mg m^{-2} hr^{-1} and g m^{-2} day^{-1} for better clarity of the magnitude of effluxes. The IT proved to have the highest while G had the lowest soil moisture throughout the study period ($r = -0.26$, Table 6.2). Autumn received the most precipitation as indicated by the high soil moisture while spring was the driest period ($r = +0.52$; Table 6.2). The winter season actually received more precipitation than summer in this study. When analysed across seasons, soil moisture had a very strong positive effect on NCER ($r = +0.9$ at $p = 0.0001$, Table 6.2).

Seasonal soil temperatures were consistently high in the DT throughout the study, while IT and G temperatures fluctuated. The DT, IT and G had consistently high, low to negligible and moderate to high penetration resistance respectively ($r = +0.36$). Spring had the hardest soils with the general order of soil hardness being Spring > Summer > Winter> Autumn resistance ($r = -0.15$); with seasonal penetration resistance having a significant negative overall effect on NCER ($r = -0.66$ at $p = 0.002$).

This above combinations of soil conditions resulted in the DT having lower NCER values in Spring (very dry + hard soils) and Summer (very hot, dry + hard soils), while IT had lower effluxes in Winter (cool, wet + soft soils); and autumn (very wet, hot + soft soils) having consistently high effluxes at all sites. The G had more or less moderate effluxes compared with the other two sites from winter to summer. The overall trend for CO_2 effluxes was stronger when analysed across seasons as it increased in the order autumn > summer > winter > spring (r = +0.70). In principle, spring effluxes were fairly negligible while winter and summer effluxes were moderate and the autumn effluxes quite high.

Table 6.2 Correlation analysis for seasonal variability

	Land Cover	**Season**	**Moisture**	**Temp**	**Pen Rest**	**NCER**
Land Cover	-					
Season	0.00	-				
Moisture	-0.26*	+0.52***	-			
Temp	+0.05	+0.68***	-0.002	-		
Pen Rest	+0.36**	-0.15*	-0.75***	+0.18*	-	
NCER	-0.20*	+0.70***	+0.90***	+0.15*	-0.66***	-

*Marked correlations are significant at *** $p<0.001$, ** $p<0.01$ and *$p < 0.05$

Table 6.3 Variations in measured assay parameters across seasons

Season	Vegetation Cover	Moisture (%)	Soil Temperature (^oC)	Penetration Resistance (Kgm^{-2})	NCER ($\mu molsm^{-2} sec^{-1}$)	NCER ($mg\ m^{-2}\ hr^{-1}$)	NCER ($gm^{-2}\ day^{-1}$)
Winter	IT	10.4	17	3.7	0.71	112.1	2.7
	DT	9.5	25.7	4.5	0.89	141.7	3.4
	G	8.5	21.6	3.8	0.80	126.7	3.0
	Average	**9.5**	**21.4**	**4**	**0.8**	**126.8**	**3.0**
Spring	IT	5.7	29	4.1	0.64	100.7	2.4
	DT	3.1	29.5	11.9	0.37	58.9	1.4
	G	3.0	23.7	9.2	0.52	82.1	2.0
	Average	**3.9**	**27.4**	**8.4**	**0.51**	**80.5**	**1.9**
Summer	IT	8.5	34	3.6	1.15	182	4.4
	DT	6.9	38	7.6	0.59	92.8	2.2
	G	5.4	34	7.2	0.87	137.8	3.3
	Average	**6.9**	**35.3**	**6.1**	**0.87**	**137.5**	**3.3**
Autumn	IT	16.2	27.1	3.2	1.98	313.3	7.5
	DT	16.2	33.5	3.7	1.50	236.9	5.7
	G	13.1	30.6	3.8	1.39	219.9	5.3
	Average	**15.2**	**30.4**	**3.6**	**1.62**	**256.7**	**6.2**

6.5 Discussion

It can be observed that soil net carbon dioxide exchange is not controlled by one factor alone but a combination of many soil variables. The fact that the study area received both winter and summer rainfall further compounded the trend. The moisture effects and its interaction with penetration resistance are clear, with CO_2 effluxes generally increasing as soil moisture increased and penetration resistance decreased. Thus CO_2 responded well to increased moisture across all vegetation. Nsabimana *et al*., (2009) also found soil CO_2 efflux rates from all species stands under study to be highest in the wet seasons (4.6–5 µmols $m^{-2}sec^{-1}$) and lowest in dry seasons (2.12µmols $m^{-2}sec^{-1}$). McLain and Martens, (2006) observed that CO_2 efflux closely followed rainfall patterns, with the lowest effluxes measured following periods of limited precipitation during the warm season.

Even though temperature had no overall significant effect on gas effluxes, its interaction with moisture and penetration resistance between seasons cannot be ignored. Nsambina *et al*., (2009) explained that in the tropics and south of it, soil temperature had a weak influence on respiration, which was related to small temperature variability throughout the year. In contrast, research in boreal and temperate regions has shown a strong association between soil CO_2 efflux and soil temperature because high temperature stimulates plant and microbial activity in these areas. In the present study, the winter months of June to August characterized by low soil temperature (16-30^{o}C) and high soil moisture (up to 16 % gravimetric moisture) had more CO_2 effluxes (up to 1.2 µmols $m^{-2}sec^{-1}$) at the G and DT than the IT (about 0.7µmols $m^{-2}sec^{-1}$). Soil penetration resistance was low (1 to 4 Kg/cm^2) during this period at all sites, hence the high gas effluxes in DT and G soils. This trend continued in September even though moisture had considerably reduced by then (~ 2 %). The spring months of October-November, and summer months of December-January; characterized by drier soil (< 10 % moisture) and hotter temperature (up to 40^{o}C) saw higher CO_2 effluxes in the IT (up to 0.8µmols $m^{-2}sec^{-1}$) than G and DT (< 0.6µmols $m^{-2}sec^{-1}$). The reduced canopy cover in DT resulted in greater heating and drying of surface soil than in the shaded IT, thus causing moisture to become a limiting factor for respiration in spite of the favourable temperatures. Moreover, the shading and higher organic C content of IT soils may have slowed down moisture loss following a precipitation event; while lack of shading and low C availability would have increased evaporative drying in the open areas in response to increased temperatures, thereby reducing the net CO_2 flux (McLain and Martens, 2006) in spring and summer.

The amount of plant biomass and litter fall could have a bearing on soil respiration rates. The IT was characterized by a thick mix of woody canopy trees and multi-stemmed succulent shrubs; while the DT and G were more open comprising mostly grasses and a few herbs (Chapter 4.4). Samawoto *et al*., (2000) found soil respiration in a regenerating forest to be in the same range as in an intact forest but significantly higher than in a treeless site that had lost its cover through burning. They attributed the decrease in soil respiration to the loss of tree root respiration at the burnt site. In our case; the reduction in vegetation at the DT was due to browsing and grazing by game animals. Zimmerman *et al*., (2009) in their study to manipulate substrate supply for soil respiration found mean total soil respiration in a 15 cm thick litter layer of a tropical rainforest to be higher (3.22 µmols $m^{-2}sec^{-1}$), than in a litter-free layer (2.02 µmols $m^{-2}sec^{-1}$) indicating that 37% of total respiration was derived from litter. Luo and Zhou, (2006) stated that removal of a substantial portion of the canopy caused soil to receive more incoming shortwave radiation during the day while trapping less long-wave radiation during the night causing higher daytime temperatures; lower night temperatures and a drier upper soil layer. Thus high temperatures and low moisture in DT and G soils increased penetration resistance (up to 14 Kg cm^{-2} in spring) due to soil hardening; resulting in limited gas emissions. The IT site's penetration resistance (< 5 Kg cm^{-2}) did not appear to be affected by dry temperatures. This could be due to higher organic carbon at this site (up to 3.4 %;) causing improved aggregation of soil hence better aeration for more respiration especially in the top layer. The February-March period experienced high temperatures (up to $40^{o}C$) and high moisture (up to 16 %) both of which raised respiration rates at all three sites, more so in the IT (about 2µmols $m^{-2}sec^{-1}$). It can be concluded that the grassland and degraded effluxes responded more to increased temperature in the winter months of June and July; while intact thicket contributed more to effluxes as temperatures increased in November-December and all three sites respired more in response to higher temperatures in February.

On the whole, high temperatures-low moisture enhanced high penetration resistance at the degraded and grassland thus grossly reducing their spring CO_2 effluxes. When investigating controls of CO_2 efflux at a Mediterranean site, Rey *et al*., (2002) found that soil respiration was controlled by temperature under moist conditions (at least 20% volumetric moisture) but was limited by soil moisture during dry periods. This shows that higher soil respiration rates in response to temperature increases may be constrained by low moisture in semi-arid soils (McLain and Martens, 2006). In their study Nsabimana *et al*., (2009) also found the relationship between soil temperature and soil CO_2 efflux not to be that strong; with temperature variation only accounting for 6–17% of seasonal CO_2 effluxes. High temperature and high moisture increased effluxes at all sites (February-March period), while low temperature and high moisture (June-July winter period) caused more effluxes in the degraded and grassland, due to reduced penetration resistance. The season followed the trend of low winter effluxes, rising spring effluxes then very high summer to autumn effluxes at the IT. The degraded and grassland had high winter, low spring then rising summer to autumn CO_2 effluxes.

Initially, it was deemed that since the IT had more vegetation and sequestered higher levels of C of above 3%; it would emit less CO_2 than the sparse DT. Our results show that net CO_2 exchange is a more complex process that is influenced by many intrinsic soil factors. In their study, Mills *et al.*, (2011) noted that over-utilisation of rangeland over time had reduced soil respiration primarily by reducing plant productivity as a result of higher animal stocking densities. They explained that plant productivity was positively correlated with root respiration and the rate of organic matter input into the soil. In this case, vegetation at the DT had been severely damaged and reduced by browsing elephants and rhino. Sundarapandian and Kirthiga, (2011) also observed that plant diversity altered the rate of soil respiration with greater soil respiration being recorded in scrubland (that had more plant diversity) compared to grassland and *Acacia* monocultures. They attributed this difference in soil respiration to plant mediated effects on the soil microclimate. In the present study, there was no overall significant difference in respiration rates among sites; but inter-seasonal variations were clear with the DT sometimes producing more effluxes (winter), IT having higher summer fluxes and G being mostly moderate. Such complexities were a result of interactions between variations in vegetation, moisture, temperature and soil physico - chemical properties between sites.

6.6 Conclusion

Soil respiration is a useful measure for detecting changes in ecosystem structure and function. It was observed that a change in the composition of plant species induced by degradation altered soil respiration rates and decreased the carbon sequestration potential of the ecosystem. Thus the degradation of thicket vegetation through animal activity resulted in creation of hard dry soils under limiting moisture conditions resulting in suppression of net respiration; but caused accelerated respiration under unlimited moisture supply. This resulted in major variations in effluxes between IT and DT vegetation. The study showed that athough semi-arid ecosystems are often limited by severe drought and their characteristic sandy soils; they can still contribute to atmospheric CO_2 emissions. Moisture, penetration resistance and temperature as well as their interaction all played a significant role in net soil respiration, with high moisture increasing respiration rate while high penetration resistance decreased it. Soil respiration was controlled by temperature when moisture supply was unlimited but during dry periods, it was limited by soil moisture. Increase in moisture is desirable since it increases crop productivity. However, moisture increase accompanied by a simultaneous rise in temperature results in accelerated respiration rates.

It can be concluded that the DT had high effluxes in winter and reduced CO_2 emissions in summer; while the opposite was true for the IT. All the cover types had minimal CO_2 effluxes in spring and accelerated CO_2 emissions in autumn. The grassland was a fairly moderate source or sink of CO_2 in most seasons compared to the other two covers. On the whole, all three cover types had moderate effluxes ranging from 3.1 – 3.8 g m^{-2} day^{-1}, thus they were not highly threatening sources of CO_2. The recommendation would be to maintain an environment of good moisture and low-moderate temperatures (such as that in the winter period) since this reduced effluxes in at least two of the covers (IT and G). The spring condition would be the most ideal in terms of limiting overall CO_2 emissions but it is not realistic practically as the limiting conditions of low moisture and increased soil hardness would hamper plant productivity and consequently its C sequestration potential. The summer condition (hot dry soils) is not ideal especially at IT as the high litter fall serves as a ready substrate for accelerated respiration under high temperature. The litter also preserves a reserve of moisture that prevents soil hardening such that gas emission rates still remain high due to the hot temperatures. The autumn condition (very high moisture and increasing temperature) on the other hand is the worst combination as it results in accelerated gas emissions. Shifts in precipitation patterns to increased winter (than autumn or summer) rains may decrease net CO_2 effluxes of semi-arid soils. Efforts must also be directed towards mitigating against further increases in atmospheric temperatures as they only serve to accelerate soil respiration rates under increased moisture supply.

CHAPTER 7: SYNTHESIS

7.1 Introduction

This chapter gives a summary of the different strands of the thesis. It provides an overall conclusion as well as recommendations based on the major findings of the study. The questions that arose from the study and the research gaps that still need to be filled are also highlighted. The chapter begins by describing the trends and relationship between historical rainfall variability and temporal land use or cover changes in Albany thicket ecosystems. It then brings to the fore specific amounts of biomass accumulated and carbon sequestrated in thicket biome. The use of C and N isotopes as a measure of land-use history and soil organic matter turnover in thicket soils are also explored. Finally, the amount of carbon dioxide effluxes released from soil under thicket at various levels of degradation is elucidated. As a conclusion, the cross-linkages between the various research questions, objectives and findings are established and recommendations made on the way forward regarding future research directions.

7.2 Thicket vegetation response to historic rainfall variability

Annual rainfall has been decreasing over the years at Grahamstown, Uitenhage and Bathurst but not at Amakhala reserve, Port Elizabeth and Port Alfred. The pooled station data however showed a general declining trend in rainfall ($r = -0.41$ at $p < 0.05$). Rainfall was highest in the 1970s at most stations, then decreased significantly in the 1980s and 1990s, a trend that has been supported by other studies. Blignaut, (2009) observed South Africa to be drier between 1997 and 2006 compared to the 1970s in eight out of nine provinces studied. This same trend was also observed for seasonal rainfall with autumn, winter and summer rainfall declining sharply in the 1980s and 1990s. Nicholson (2000) reported that rainfall variability over east and southern Africa was mostly influenced by the El Niño Southern Oscillation (ENSO) cycle and sea surface temperatures (SSTs) over the Atlantic and Indian Oceans. This results in Southern Africa rainfall to be affected by drier than normal conditions during El Niño and wetter than normal conditions during La Niña events (Philippon *et al*. 2011). Both the frequency and magnitude of wet events contributed to annual rainfall as seen from the significant positive correlations between the rainfall classes of 10 mm and above with the daily rainfall index (DI) and annual rainfall. The frequency and length of dry spells however did not show any significant relations with annual rainfall or DI for most stations, with dry spells only showing an increasing trend at Bathurst. This showed that wet rather than dry spells had more impact on rainfall in this study.

Land use / cover (LUC) change analysis showed than the Albany study area is dominated by over 30 % transformed vegetation. There has been an increase in the area occupied by transformed and degraded vegetation over the years, with the 1989-1999 period being mostly responsible for this change. Thus the area occupied by degraded land increased from 9 % (76 000 ha) in 1989 to 17.5 % (144 743 ha) in 2009 while that occupied by transformed vegetation increased from 33 % (264 000ha) in 1989 to 34 % (287 965 ha) in 2009. The loss of vegetation cover was attributed to heavy browsing by wildlife, and overgrazing by domestic herbivores like goats especially in the communal areas. Intact vegetation decreased significantly by 17.4 % (144 094 ha) in 1989-1999 but it regained its aerial coverage between 1999 and 2009. Farmland however continued to decline over the years and had an overall loss of 1.8 % (16 125 ha) by 2009. Land use-change to game ranching in commercial and goat pastoralism in communal lands was mostly responsible for the decline in farmland. In an effort to diversify their operations private landowners within EC have been shifting from pastoralism to game ranching since the 1980s (Smith and Wilson, 2002). Communal farmers have also reduced their cultivation activities and devoted their land mostly to goat browsing. Since there was no distinction between natural and reforested area, the increase in intact vegetation between 1999 and 2009 could have been due to the growth in area covered by plantation forest in EC since 2004 (DEDEA, 2009).

It was difficult to state precisely the effect of rainfall variability on land-use change or vice-versa. This is because land-use change could be caused by a number of other factors such as unsustainable grazing, heavy browsing, land clearance and bush encroachment. The LUC change - rainfall variability analysis however showed that 1999 recorded the least rainfall and had the lowest NDVI mean, maximum and sum. The relationship was mostly significant at Grahamstown which had the lowest NDVIs in 1999 when rainfall was lowest, the highest NDVI in 1989 when rainfall was highest and moderate NDVIs in 2009 when rainfall was moderate. No clear linkages however were established at the other stations. The implication of these trends is that even though thicket species are resilient to marginal conditions of soil fertility and moisture, sustained long dry periods are not desirable for their productivity. Reports show that transformed thicket seems to be stable in the short to medium term as long as the climate is stable. However, canopy trees tend to be vulnerable to periodic droughts, that cause significant mortality in transformed compared to intact habitats (Lechmere-Oertel *et al*., 2005). As mentioned before, thicket species are very difficult to recover once destroyed, more so under conditions of limiting moisture. Since rainfall has been declining over the years in the study area, it is best to rather maintain thicket in its pristine state, to prevent its total eradication from the ecosystem. Having identified the declining rainfall trends and considerable land use/cover changes that have taken place, it was then necessary to establish the implications of such trends for C and CO_2 flux; and to ascertain if ^{13}C and ^{15}N isotope ratios would be indicators of the land use/cover changes identified.

7.3 Plant and soil C sequestration under thicket

The soil organic C sequestered under intact thicket was quite high, reaching levels of 3.4 % (i.e. 3.4 t C / ha of land) in the top 10 cm then decreasing with depth. Upon degradation however soil organic C in thicket decreased to moderate levels of (1.1-1.3 %) and was very low in grassland (≤ 0.5 % C). The species inventory done at the study site found a diversity of growth forms under intact thicket (IT) that included canopy trees, woody shrubs, succulent shrubs and ephemerals. All the woody species had however disappeared under degraded thicket (DT) to be replaced by the invasive species *Pteronia incana* (blue bush), some grasses and herbs. The grassland (G) had a predominance of the grasses *Themeda triandra* and *Panicum maximum* and a few herbs. The disappearance of woody and succulent thicket species upon degradation was prolific; resulting in the creation of an unstable desert-like system dominated by annual grasses, forbs and dwarf shrubs. Because of this varied diversity of species at the different sites, the plant biomass and C sequestered also differed immensely. Above ground biomass reached exceptionally high levels of 330 000 kg/ha in IT; but was only 22 000 kg/ha in DT and as low as 6 700 kg/ha in G vegetation. This same trend was maintained for biomass carbon with values of 158 000, 10 600 and 3 200 kg/ha respectively. The study confirmed the high floristic diversity and C sequestration potential of thicket vegetation in an undisturbed state. In all, the conversion of IT to a DT resulted in a net loss of 147 400 kg/ha of biomass C and 12 000 kg/ha of soil organic C. Over-browsing by game animals was quite evident at the DT; as it was often characterised by several animal tracks that gave way to very hard exposed soils with high bulk densities (mostly above 1 g cm^{-3}) and penetration resistance (up to 14 Kgm^{-2}). Thicket degradation also resulted in creation of an unfavourable soil environment prone to dispersion, crusting, erosion and accelerated runoff.

This had a tremendous effect on productivity of thicket species such that their recovery was highly suppressed, thus making way for invasive plant species and weak ephemerals. C sequestration under grassland was low; due to the poor nutritional level of the soil (i.e. > 70 % sand soils with low exchangeable bases and nitrogen (≤ 0.06 %). For this reason, it is recommended that succulent thicket be maintained in its pristine condition as its recovery upon degradation is difficult. In their natural state, thicket plants are very resilient and tolerant to marginal conditions of soil moisture and nutrition. Re-establishing them however under these same conditions is difficult since they have poor seedling recruitment and low seed dispersal ability.

7.4 ^{13}C and ^{15}N isotope ratios as indicators of land-use change and soil OM turnover

Soil C isotope depletion showed that the original vegetation in the IT had intermediate photosynthesis between C_3 and CAM metabolism; while DT had a predominance of CAM plants as it exhibited more positive $\delta\ ^{13}C$ in soil averaging -15.81 ‰; and the G soil showed a predominance of C_4 plants in original vegetation. Major shifts in metabolism were observed in thicket species however with most of them leaning towards C_3 photosynthesis. This shift was more substantial at DT reflecting a greater change in species composition at this site over the years. The G however showed little variation in isotope ratios between the soil and current vegetation, indicating that plants at this site mostly maintained their original C_4 metabolism. C isotope enrichment ratios were highest at DT (-9.84‰) and lowest at G (+1.5‰) further confirming greater changes in vegetation at DT. Reports of increase in C_3 shrubs in an area due to long-term heavy grazing by livestock are evident in literature (Boutton *et al*., 1998). In the present study, heavy browsing by wildlife at the DT was responsible for altering predominantly CAM vegetation to C_3 shrubs and herbs. The higher increase of soil $\delta\ ^{13}C$ with increasing depth at DT also proved that the deeper soil layers with higher $\delta\ ^{13}C$ were derived from CAM dominated vegetation while the top soil layers with more negative ratios signalled a shift towards C_3 metabolism in current vegetation. ^{13}C enrichment at deeper soil depths could also have been a result of preferential decomposition of ^{13}C rich organic material by microorganisms during biological decay.

In the case of the N isotopes, soil $\delta\ ^{15}N$ was significantly higher at IT and lowest at DT ($p< 0.001$); which corresponded with higher soil N at IT. It was concluded that the IT was more enriched in N with higher N mineralization rates, which gave rise to the more positive $\delta\ ^{15}N$.

High soil N mineralization rates often correlate with denitrification or leaching of inorganic N; leading to losses of ^{15}N-depleted N, leaving behind a ^{15}N enriched N pool. The lower δ^{15}N values of plants than soils (represented by the negative ER values) was a result of assimilation of inorganic soil N depleted in ^{15}N due to microbial transformations by plants (Pardo *et al.*, 2007). Soil N becomes depleted in ^{15}N because microbes discriminate against the heavier ^{15}N during microbial transformations of N in the soil. The plant δ^{15}N values typically reflected the δ^{15}N of the inorganic N they take up. Thus, the trend for foliar δ^{15}N was similar to that of soil δ^{15}N, with the IT having the most positive δ^{15}N values while DT had lower values. Since litter inputs are strong determinants of the δ^{15}N of surface soil horizons; ^{15}N-depleted litter will lower the δ^{15}N of the organic soil horizons, while ^{15}N-enriched litter will lead to ^{15}N enriched surface horizons (Pardo *et al.*, 2007). ^{15}N also showed enrichment with increasing soil depth, serving to prove that the organic matter inputs from the current thicket vegetation are not in isotopic equilibrium with soil organic carbon (more so at DT). Thus land that used to be dominated by CAM plants is now occupied by plants that have shifted more towards C_3 metabolism. This is not a desirable trait since C_3 plants are not as water-use efficient as CAM plants, making them more vulnerable to drier soil conditions. Under marginal fertility and moisture limitation, net primary productivity and propagation potential of degraded thicket species is diminished. This could explain the poor above-ground biomass and low recovery rate at DT. More effort needs to be directed towards restoring the vegetation to its original form since this ensures better water-use efficiency by the plant.

7.5 The impact of land-use / cover type and soil properties on CO_2 effluxes from the soil

The results showed that net carbon dioxide exchange in the soil was controlled by a combination of many soil variables, so it was not possible to pin-point one factor in isolation and infer its effect on effluxes. Moisture and penetration resistance showed a good interactive effect; so as moisture increased, soils became softer due to lower penetration resistance; causing increased CO_2 effluxes from the soil. Soil temperature effects that appeared insignificant in monthly comparisons were more apparent across seasons and in interaction with other variables. Thus the cold winter months characterized by low soil temperature (16-30^{o}C), high soil moisture (up to 16 % gravimetric moisture) and low penetration resistance (1 to 4 Kg/cm^2) had more CO_2 effluxes (up to 1.2 µmol m^{-2}sec^{-1}) at the G and DT than the IT (about 0.7µmols m^{-2}sec^{-1}). On the other hand; spring and summer months with drier (< 10 % moisture), harder and hotter soils (up to 40^{o}C), had higher CO_2 fluxes in the IT (up to 0.8µmols m^{-2}sec^{-1}) than G and DT (< 0.6µmols m^{-2}sec^{-1}). The type of canopy cover also had an influence on effluxes. As a result, the DT due to a lack of canopy cover experienced greater heating and drying of surface soil than the shaded IT, causing moisture to become a limiting factor for respiration at DT despite the favourable summer temperatures. The higher plant biomass and litter fall at IT served as ready substrate contributing to accelerated respiration rates than at DT and G that had lower biomass. This could explain the higher CO_2 effluxes at IT in most of the seasons. Since soil respiration comprises root and microbial respiration, the higher species diversity as IT could have resulted in higher root respiration causing higher effluxes at this site. It was initially believed that since IT had more above-ground biomass and greater C sequestration ability than DT and G; it would result in less CO_2 emissions from the soil. This proved not to be the case. The higher organic C at IT actually resulted in soils with good aggregation; which were consistently soft throughout the study causing unrestricted effluxes. On the other hand, over-utilisation of DT over time caused reduced plant productivity thereby hampering soil respiration activity. However, there was no overall significant difference in respiration rates among sites in this study. What came out clearly were the inter-seasonal variations with the DT sometimes producing more effluxes (winter), IT having higher summer fluxes and G being mostly moderate.

In conclusion, semi-arid ecosystems, though often limited by severe drought and their characteristic sandy soils still have a role to play in CO_2 emissions into the atmosphere. It was observed that a change in the composition of plant species induced by degradation altered soil respiration rates and decreased the carbon sequestration potential of thicket. Thicket degradation through animal activity resulted in creation of hard dry soils under limiting moisture conditions with suppressed net respiration; but had accelerated respiration under unlimited moisture supply. High soil moisture is desirable since it increases plant productivity. However a simultaneous increase of both moisture and temperature is not ideal since it causes accelerated soil respiration rates. It was observed that an environment of good moisture and low-moderate temperatures (such as that in the winter period) was more favourable since it enhanced good plant productivity with moderate amounts of CO_2 emissions from the soil. Mitigating efforts against any further increases in atmospheric temperatures must therefore be promoted to reduce accelerated soil respiration rates under increased moisture supply. This can be done through minimizing anthropogenic activities such as fossil fuel combustion, cement production, deforestation and land use changes that enhance emission of CO_2 into the atmosphere as they cause increased warming.

7.6 Overall conclusion and recommendations

As observed in the study, there has been a general decline in rainfall and increased transformation and degradation of vegetation over the years in the Albany thicket region. This transformation is mainly attributed to land-use change to game ranching and goat pastoralism, with uncontrolled browsing activities causing destruction of thicket species. The decline in rainfall, which is mostly attributed to the El Niño Southern Oscillation cycle (ENSO) episodes, only makes plants in transformed thicket more vulnerable to degradation as their recovery is hampered by a limited moisture supply under degradation. Succulent thicket vegetation, in its natural state is water-use efficient and effective at maintaining a healthy ecosystem in semi-arid environments, through accumulation of high biomass and sequestering large amounts of C. It is resilient to marginal conditions of fertility and moisture, and adapts to this by developing thorny succulent species that store large reserves of moisture. Thicket species adapt to limited moisture by having diverse CO_2 fixation patterns, which switch between C_3 photosynthesis during the day and CAM metabolism at night. Through CAM metabolism, carbon dioxide is assimilated at night when potential evapotranspiration is lowest, thus the amount of water lost per unit of carbon gained is relatively low making it a good mechanism for conserving water in arid regimes. As a result, a variety of dense growth forms of canopy trees, woody shrubs, succulent shrubs and ephemerals with diverse photosynthetic pathways proliferate in undisturbed thicket. Human activity through prolonged domestic herbivory, cultivation and land-use change can bring a lot of disturbance to this balance. Soon the pristine thicket is transformed to land characterised by perennial grasses, dwarf shrubs and herbs. The canopy trees slowly disappear from the landscape with more degradation; to be replaced by bushes of invader species characterised by low phytomass and poor litter production. Again most of the CAM species that are well adapted to low moisture are reduced in numbers until the vegetation becomes dominated by C_3 plants that are less efficient at preserving water. Degraded thicket land is also characterised by soil with reduced water infiltration, has a tendency to crust and is prone to compaction and erosion. The soils become very hard when dry leading to reduced soil respiration rates which ultimately affect plant productivity. These same soils also experience accelerated respiration levels when the moisture and temperatures are high causing high carbon dioxide emissions.

Land-use change that is detrimental to proliferation of thicket species is therefore not sustainable in most semi-arid and arid environments. This is because it is very difficult to re-establish the succulent species once destroyed. The seedlings of thicket plants occur at low densities limited to the microclimate underneath the canopy hence they make little contribution to the regeneration of thicket species. Moreover, the seed dispersal mechanisms of thicket plants are often poor making regeneration impossible once destroyed. Prolonged dry conditions only serve to exacerbate recovery of thicket plants as they increase mortality of canopy species.

On the whole, the study showed significant declines in rainfall trends at Grahamstown, Uitenhage and Bathurt as well as the overall pooled data for all stations. Over 30 % of the land in the Albany thicket is transformed vegetation with degraded vegetation also increasing in area due to unsustainable browsing and intense grazing activities. This transformation of thicket has seen significant decline in biomass accumulated, and the overall C sequestration potential of thicket under degradation has decreased significantly. The isotope dilution experiment supported the change in vegetation composition as most of the plants in IT and DT showed a shift to C_3 metabolism which was in contrast to their soils that had a record of predominantly CAM metabolism. The C_3 plants proved to be less water-use efficient as they struggle to recover upon degradation. Degradation of thicket also resulted in creation of hard crusting soils which suppressed soil respiration rates under limited moisture supply and thus reduced the productive potential of thicket species. Where moisture was not limited however, respiration rates were quite high under DT with more effluxes being observed as temperatures simultaneously increased in all land covers.

The recommendations that were drawn from this study are that thicket vegetation is a good sink of carbon that should be preserved in its natural condition to optimize its carbon sequestration potential. More of the land, particularly in the communal area should be turned into conservancy areas to minimize further degradation. Where conservancies are not possible, rotational grazing should be encouraged to minimize degradation and allow species enough time to recover. The stocking densities for animals must be monitored to avoid over-grazing. This serves as a serious problem in game reserves when large herbivores are left to multiply uncontrollably in such a way that they end up causing even more damage than domestic herbivores. As mentioned before, elephants, rhino and kudu are very destructive to thicket vegetation, a situation which is worsened by their over-population. Game owners could maintain sustainable numbers through sterilisation of animals once they have produced a particular number of young ones to avoid their overpopulation. Sporadic checks should be made by the department of environmental affairs to see if game owners are maintaining sustainable animal numbers that adhere to the carrying capacity of the land. Another way of controlling animal numbers could be to introduce hunting licences for tourists, which would also help the ranchers in generating more revenue from their ecotourism.

The preservation of thicket will also indirectly curb the problems of declining rainfall and limited freshwater availability in the EC province. This is because the dense vegetation cover will minimize the impact of runoff and erosion thereby preventing siltation of freshwater bodies. Again, because of their water-use efficiency, succulent thicket plants have better resilience to limited moisture supply. The quality and quantity of freshwater must also be preserved by limiting the amount of waste material from municipal and industrial pollutants. Industries should be made to participate in cleaning up the environment by removal heavy metals and other dangerous contaminants produced in waste material. Again greenhouse gas emissions from fossil fuel burning and other industrial activities should be kept at a minimum to prevent further atmospheric temperature increases and rainfall extremities.

Though this study tried as much as possible to answer the initial research questions, some gaps still remained that need further exploration. Due to limited time and instrumentation, it was not possible to monitor emission of more greenhouse gases (GHGs) over a longer time period as initially envisioned. Thus the study only monitored CO_2 emissions over a 10 month period. Future research directions should observe effluxes of other GHGs like methane and nitrous oxide across many seasons to capture long-term variability. This could be done under other biomes such as forests, fynbos, savannah and other management regimes like cultivated land or plantations. More soil parameters such as porosity, conductivity, diffusivity and resistivity should also be included during the gas monitoring exercise. A distinction could also be made between contribution from microbial, faunal and root respiration in the soil. The rainfall trend analysis was limited by non-availability of data so it was only done for a 41 year period. Daily rainfall data in particular lacked in most stations and was characterised by a lot of missing data for some of them. The government must direct more effort in recording daily data in more stations around the country. That way it would be possible to see clearer trends of extreme weather events, since they are directly linked to natural disasters like floods, droughts and cyclones. There were limitations in the LUC change detection as satellite images could only be obtained from the 1980s onwards. Better comparisons could have been made with images from the 1970s since this was the period that experienced major climate shifts. Spot images would have been more ideal in observing LUC changes since they have better resolution. However, there were no spot images available for the area prior to 1980. More parameters such as leaf area index could also be explored in future studies to give a better understanding on variability of land cover changes over the years.

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APPENDICES

Appendix A. Rainfall variability at Amakhala

Variable	Correlations (Amakhala 1970-2010 Spreadsheet) Marked correlations are significant at p < .05000 N=40 (Casewise deletion of missing data)				
	Means	Std.Dev.	Year	Annual Tota	Z score
Year	1989.625	11.8855	1.000000	0.104279	0.104279
Annual Total	443.069	113.1190	0.104279	1.000000	1.000000
Z score	0.000	1.0000	0.104279	1.000000	1.000000

Appendix B. Rainfall variability at Port Elizabeth

Variable	Correlations (Port Elizabeth 1970-2010 Spreadsh Marked correlations are significant at p < .05000			
	Means	Std.Dev.	YEAR	Annual Tota
YEAR	1990.000	11.9791	1.000000	-0.247611
Annual Total	605.268	154.3491	-0.247611	1.000000

Appendix C. Rainfall variability at Port Alfred

Variable	Correlations (Port Alfred 1970-2010) Marked correlations are significant at p < .05000														
	Means	Std.Dev.	Year	Annual Tota	Days with no Rain	Largest dry spell	Daily index	Highest Daily Rainfa	0.1-9.9 mm	10-19 9 mm	20-29.9 mm	30-39.9 mm	40-49.9 mm	Rain> 50mm	Days > 19.9mm
Year	1990	12.0	1.00	-0.22	-0.47	-0.06	-0.51	0.02	0.58	-0.19	-0.13	-0.33	-0.24	-0.10	-0.43
Annual Total	646	170.7	-0.22	1.00	-0.30	-0.35	0.53	0.50	0.08	0.36	0.60	0.46	0.35	0.65	0.79
Days with no Rain	285	20.6	-0.47	-0.30	1.00	0.54	0.52	-0.10	-0.94	-0.03	-0.24	0.08	-0.04	-0.08	-0.00
Largest dry spell	27	10.3	-0.06	-0.35	0.54	1.00	0.27	-0.14	-0.55	-0.11	-0.12	0.03	-0.05	-0.17	-0.08
Daily index	8	2.3	-0.51	0.53	0.52	0.27	1.00	0.47	-0.71	0.30	0.23	0.51	0.32	0.57	0.65
Highest Daily Rainfa	70	31.5	0.02	0.50	-0.10	-0.14	0.47	1.00	0.02	0.13	0.03	0.09	0.06	0.67	0.18
0.1-9.9 mm	63	19.7	0.58	0.08	-0.94	-0.55	-0.71	0.02	1.00	-0.16	0.05	-0.15	-0.08	-0.05	-0.21
10-19.9 mm	11	3.7	-0.19	0.36	-0.03	-0.11	0.30	0.13	-0.16	1.00	0.20	-0.12	0.05	0.14	0.15
20-29.9 mm	4	2.0	-0.13	0.60	-0.24	-0.12	0.23	0.03	0.05	0.20	1.00	0.12	0.14	0.12	0.73
30-39.9 mm	2	1.4	-0.33	0.46	0.08	0.03	0.51	0.09	-0.15	-0.12	0.12	1.00	0.11	0.42	0.61
40-49.9 mm	1	1.0	-0.24	0.35	-0.04	-0.05	0.32	0.06	-0.08	0.05	0.14	0.11	1.00	0.09	0.48
Rain> 50mm	1	1.0	-0.10	0.65	-0.08	-0.17	0.57	0.67	-0.05	0.14	0.12	0.42	0.09	1.00	0.48
Days > 19.9mm	7	3.4	-0.43	0.79	-0.00	-0.08	0.65	0.18	-0.21	0.15	0.73	0.61	0.48	0.48	1.00

Appendix D. Rainfall variability at Bathurst

Variable	Correlations (Barthust 1970-2001) Marked correlations are significant at p < 0.05													
	Mean	Std Dev	Year	Annual Total	Days with no rain	Largest dry spell	Daily index	Highest daily rainfall	0.1-9.9mm	10-19.9mm	20-29.9mm	30-39.9mm	40-49.9mm	>50mm
Year	1985	9.4	-											
Annual Total	731	167.8	-0.32	-										
Days with no rain	263	9.1	0.32	-0.33	-									
Largest dry spell	23	7.3	0.41	-0.28	0.17	-								
Daily index	7.2	1.6	-0.19	0.91	0.06	-0.22	-							
Highest daily rainfall	83.3	40.2	-0.11	0.68	0.16	-0.07	0.77	-						
0.1-9.9mm	81.8	8.6	-0.25	-0.11	-0.81	0.02	-0.46	-0.32	-					
10-19.9mm	11.9	4.1	-0.04	0.31	-0.32	-0.29	0.20	0.06	-0.16	-				
20-29.9mm	4.3	1.8	0.03	0.06	-0.32	-0.00	-0.08	-0.32	0.23	-0.13	-			
30-39.9mm	1.5	1.5	-0.04	0.62	-0.10	-0.16	0.66	0.21	-0.32	0.33	-0.07	-		
40-49.9mm	0.9	0.7	-0.09	0.37	0.05	-0.01	0.44	0.24	-0.23	0.08	-0.24	0.38	-	
>50mm	1.9	1.5	-0.38	0.82	-0.07	-0.21	0.82	0.72	-0.12	-0.11	-0.12	0.37	0.25	-
>19.9mm	8.6	3.2	-0.21	0.81	-0.26	-0.19	0.76	0.31	-0.13	0.05	0.44	0.71	0.38	0.65

Appendix E. Rainfall variability at Uitenhage

Variable	Correlations (Uitenhage 1970-2010) Marked correlations are significant at p < 0.05													
	Mean	Std Dev	Year	Annual Total	Days with no rain	Largest dry spell	Daily index	Highest daily rainfall	0.1-9.9mm	10-19.9mm	20-29.9mm	30-39.9mm	40-49.9mm	>50mm
Year	1990	12	-											
Annual Total	471	142.2	-0.32	-										
Days with no rain	302	20.6	-0.74	-0.11	-									
Largest dry spell	35	14.3	-0.43	-0.11	0.66	-								
Daily index	8	3.6	-0.72	0.59	0.67	0.49	-							
Highest daily rainfall	58	25.2	-0.14	0.72	-0.07	0.08	0.51	-						
0.1-9.9mm	50	22	0.82	-0.08	-0.98	-0.62	-0.77	-0.04	-					
10-19.9mm	8	2.9	-0.51	0.43	0.24	0.10	0.42	0.26	-0.39	-				
20-29.9mm	2	1.6	-0.32	0.29	0.08	-0.09	0.19	-0.04	-0.17	0.10	-			
30-39.9mm	1	1.4	-0.39	0.66	0.16	0.06	0.52	0.31	-0.27	0.15	0.13	-		
40-49.9mm	1	0.8	-0.11	0.52	-0.11	-0.12	0.32	0.17	0.01	0.21	0.08	0.30	-	
>50mm	1	1.2	-0.14	0.75	0.03	0.03	0.59	0.81	-0.14	0.15	-0.07	0.40	0.25	-
>19.9mm	6	3.0	-0.43	0.87	0.10	-0.04	0.64	0.48	-0.26	0.23	0.57	0.75	0.52	0.60

Appendix F. Rainfall variability at Grahamstown

	Correlations (Grahamstown 1970-2010) Marked correlations are significant at p < .05000														
Variable	Means	Std. Dev.	Year	Annua Total	Days with no Rain	Largest dry spell	Daily index	Highest Daily Rainfall	0.1-9.9 mm	10-19.9 mm	20-29.9 mm	30-39.9 mm	40-49.9 mm	> 50mm	> 19.9 mm
Year	1990	12.0	1.00	-0.59	-0.16	-0.07	-0.57	-0.47	0.36	-0.51	-0.18	-0.36	-0.37	-0.42	-0.39
Annual Total	649	206.4	-0.59	1.00	-0.19	-0.20	0.70	0.70	-0.11	0.66	0.17	0.56	0.60	0.74	0.69
Days with no Rain	275	18.4	-0.16	-0.19	1.00	0.77	0.53	0.11	-0.95	0.04	0.01	-0.23	0.10	-0.19	-0.15
Largest dry spell	29	15.1	-0.07	-0.20	0.77	1.00	0.43	-0.03	-0.75	0.19	-0.06	-0.17	-0.03	-0.17	-0.15
Daily index	7	2.6	-0.57	0.70	0.53	0.43	1.00	0.67	-0.74	0.57	0.15	0.27	0.58	0.49	0.62
Highest Daily Rainfa	68	38.6	-0.47	0.70	0.11	-0.03	0.67	1.00	-0.24	0.20	0.06	0.09	0.55	0.67	0.38
0.1-9.9 mm	71	19.2	0.36	-0.11	-0.95	-0.75	-0.74	-0.24	1.00	-0.32	-0.10	0.05	-0.24	0.02	-0.11
10-19.9 mm	12	4.3	-0.51	0.66	0.04	0.19	0.57	0.20	-0.32	1.00	0.01	0.38	0.20	0.30	0.39
20-29.9 mm	4	1.6	-0.18	0.17	0.01	-0.06	0.15	0.06	-0.10	0.01	1.00	0.07	0.08	-0.19	0.51
30-39.9 mm	1	1.4	-0.36	0.56	-0.23	-0.17	0.27	0.09	0.05	0.38	0.07	1.00	0.02	0.16	0.58
40-49.9 mm	1	1.0	-0.37	0.60	0.10	-0.03	0.58	0.55	-0.24	0.20	0.08	0.02	1.00	0.45	0.58
> 50mm	1	1.4	-0.42	0.74	-0.19	-0.17	0.49	0.67	0.02	0.30	-0.19	0.16	0.45	1.00	0.50
> 19.9mm	7	3.4	-0.39	0.69	-0.15	-0.15	0.62	0.38	-0.11	0.39	0.51	0.58	0.58	0.50	1.00

Appendix G. Rainfall comparisons between stations

	Correlations (Rainfall Station Comparisons) Marked correlations are significant at p < .05000								
Variable	Means	Std.Dev.	Year	Amakhala	Port Elizabeth	Grahamstown	Port Alfred	Uitenhage	Bathurst
Year	1990.0	12.0	1.00	0.10	-0.25	-0.59	-0.22	-0.32	-0.32
Amakhala	443.1	113.1	0.10	1.00	0.62	0.44	0.40	0.61	0.52
Port Elizabeth	605.3	154.3	-0.25	0.62	1.00	0.59	0.71	0.74	0.66
Grahamstown	649.4	206.4	-0.59	0.44	0.59	1.00	0.57	0.66	0.78
Port Alfred	645.7	170.7	-0.22	0.40	0.71	0.57	1.00	0.66	0.81
Uitenhage	471.2	142.2	-0.32	0.61	0.74	0.66	0.66	1.00	0.75
Bathurst	731.5	167.8	-0.32	0.52	0.66	0.78	0.81	0.75	1.00

Appendix H. A two-way analysis of variance of soil ^{13}C isotope depletion

Variate: Soil ^{13}C isotope

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Point stratum	4	23.598	5.900	5.41	
Point.*Units* stratum					
Land cover	2	419.924	209.962	192.70	<.001
Depth	2	37.102	18.551	17.03	<.001
Land cover.Depth	4	23.820	5.955	5.47	0.002
Residual	32	34.867	1.090		
Total	44	539.312			

Appendix I. A two-way analysis of variance of soil ^{15}N isotope depletion

Variate: Soil ^{15}N isotope

Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Point stratum	4	0.5441	0.1360	0.64	
Point.*Units* stratum					
Land cover	2	17.5329	8.7665	41.55	<.001
Depth	2	2.6130	1.3065	6.19	0.005
Land cover.Depth	4	2.8438	0.7110	3.37	0.021
Residual	32	6.7515	0.2110		
Total	44	30.2853			

Appendix J. A two-way analysis of variance of plant ^{13}C isotope depletion

Variate: Plant ^{13}C

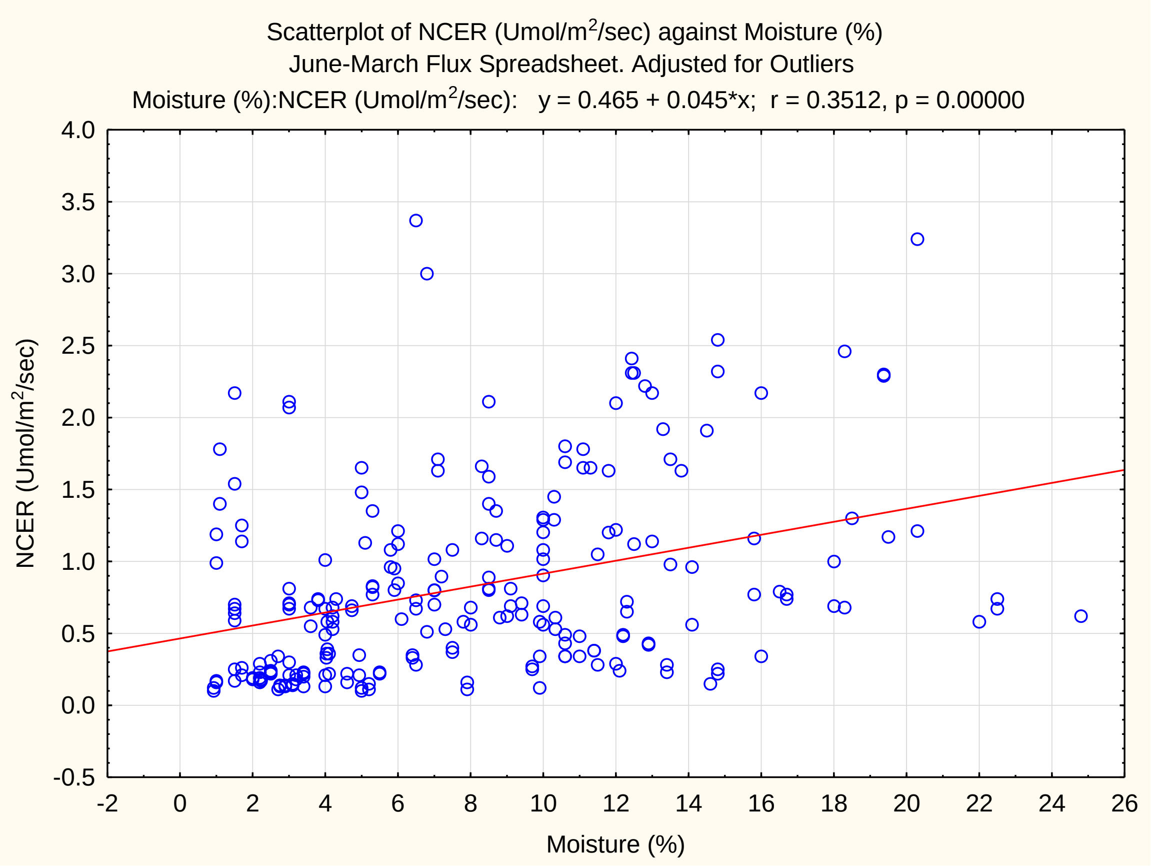
Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Biome	2	364.00	182.00	8.57	0.002
Residual	23	488.27	21.23		
Total	25	852.27			

Appendix K. A two-way analysis of variance of plant ^{15}N isotope depletion

Variate: ^{15}N

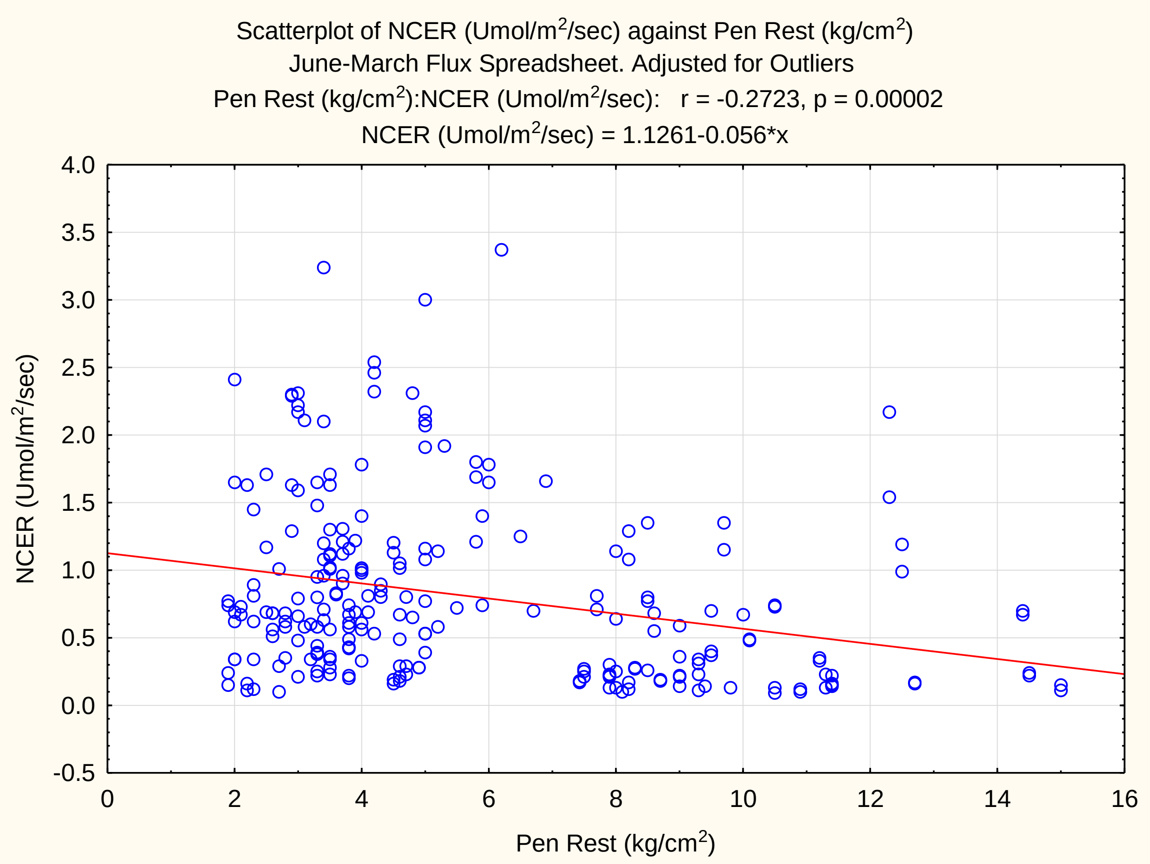
Source of variation	d.f.	s.s.	m.s.	v.r.	F pr.
Biome	2	8.059	4.030	1.08	0.357
Residual	23	86.063	3.742		
Total	25	94.122			

Appendix L. Scatter Plot of NCER against soil moisture



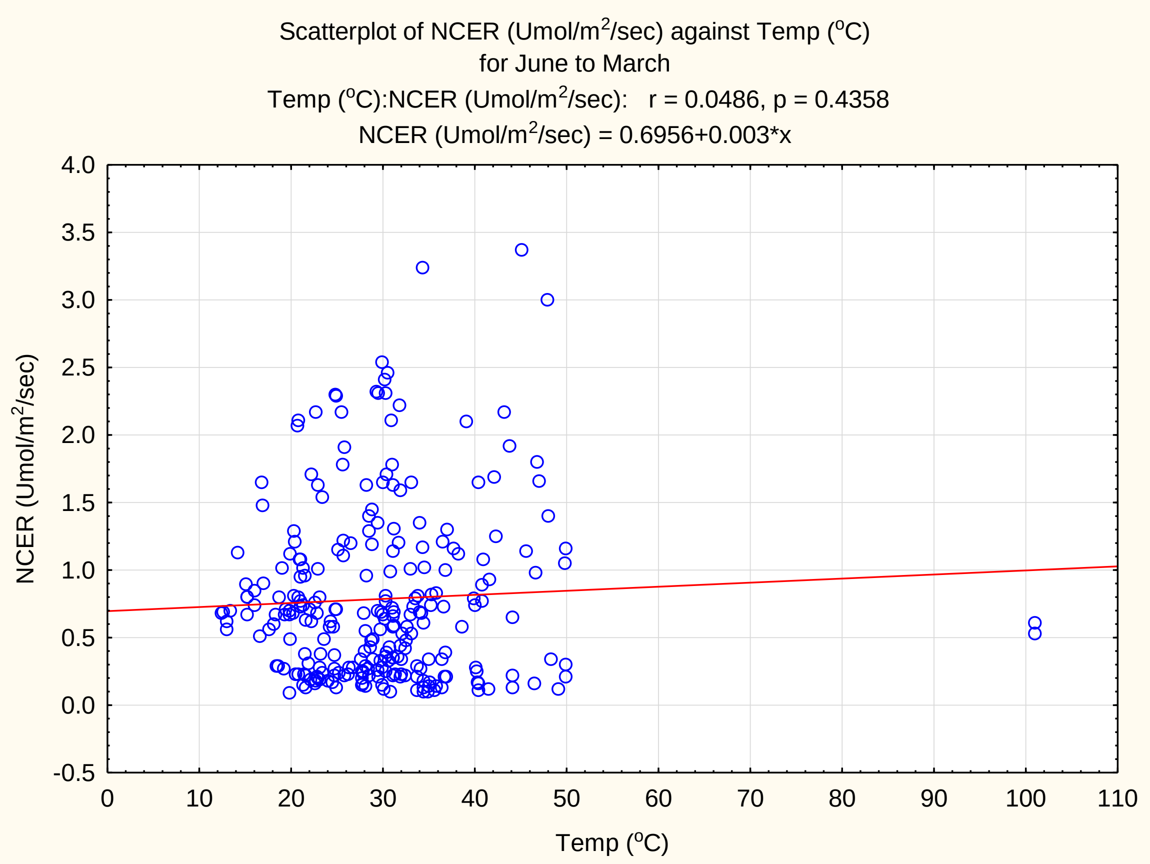
*Graph shows regression equation, correlation coefficient (r) and p value

Appendix M. Scatter Plot of NCER against soil penetration resistance



Scatterplot of NCER (Umol/m^2/sec) against Pen Rest (kg/cm^2)
June-March Flux Spreadsheet. Adjusted for Outliers
Pen Rest (kg/cm^2):NCER (Umol/m^2/sec): r = -0.2723, p = 0.00002
NCER (Umol/m^2/sec) = 1.1261-0.056*x

Appendix N. Scatter Plot of NCER against soil temperature



Scatterplot of NCER (Umol/m^2/sec) against Temp (^{o}C)
for June to March
Temp (^{o}C):NCER (Umol/m^2/sec): r = 0.0486, p = 0.4358
NCER (Umol/m^2/sec) = 0.6956+0.003*x

Appendix O. Regression analysis of NCER against other measured variables

Response variate: NCER
Fitted terms: Constant, Moisture, PenRest, Temp, Q, U, dCO2, Cref

Summary of analysis

Source	**d.f.**	**s.s.**	**m.s.**	**v.r.**	**F pr.**
Regression	7	120.492	17.213172	3382.01	<.001
Residual	207	1.054	0.005090		
Total	214	121.546	0.567971		

Percentage variance accounted for 99.1
Standard error of observations is estimated to be 0.0713.

Estimates of parameters

Parameter	**estimate**	**s.e.**	**t(207)**	**t pr.**
Constant	0.778	0.213	3.65	<.001
Moisture	0.00553	0.00113	4.90	<.001
PenRest	0.00375	0.00185	2.02	0.044
Temp	0.000349	0.000821	0.43	0.671
Q	0.0000105	0.0000627	0.17	0.867
U	0.000258	0.000173	1.49	0.137
dCO2	0.17189	0.00126	136.24	<.001
Cref	-0.0630	0.0128	-4.92	<.001