LAND USE/COVER CHANGE MODELLING AND LAND DEGRADATION ASSESSMENT IN THE KEISKAMMA CATCHMENT USING REMOTE SENSING AND GIS.

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Submitted in fulfillment of the requirement for the degree of

PHILOSOPHIAE DOCTOR

in the Faculty of Science
at the Nelson Mandela Metropolitan University

January 2011

Promoter: Professor Vincent Kakembo

AUTHOR'S DECLARATION

I hereby declare that the work contained in this thesis is my own original work and that I have not previously submitted it, in its entirety or in part, at any university for a degree.

Abstract

Land degradation in most communal parts of the Keiskamma catchment has reached alarming proportions. The Keiskamma catchment is particularly predisposed to severe land degradation associated with soil erosion, thicket degradation and deteriorating riparian vegetation. There is a close coupling between land use/cover dynamics and degradation trends witnessed in the catchment. Soil erosion is prevalent in most of the communal areas in the catchment. The principal aim of this study was to investigate land use/cover trends, model the spatial patterns of soil loss and predict future land use/cover scenarios as a means of assessing land degradation in the Keiskamma catchment.

Multi-temporal Landsat satellite imagery from 1972 to 2006 was used for land use/cover change analyses using object-oriented post-classification comparison. Fragmentation analysis was performed by computing and analyzing landscape metrics in the riparian and adjacent hillslope areas to determine the land cover structural changes that have occurred since 1972. The landscape function analysis was used to validate the current rangeland conditions in the communal areas and the former commercial farms. The current condition of the riparian zones and proximal hillslopes was assessed using the Rapid Appraisal of Riparian Condition and future land use/cover scenarios were simulated using the Markov-cellular automata model. Spatial patterns of soil loss in the Keiskamma catchment were determined using the Sediment Assessment Tool for Effective Erosion Control (SATEEC), which is a GIS based RUSLE model that integrates sediment delivery ratios. Object oriented classification was used to map soil erosion surfaces and valley infill in ephemeral stream channels as a means of demonstrating the major sediment transfer processes operating in the Keiskamma catchment. The Mahalanobis distance method was used to compute the topographic thresholds for gully erosion. To understand the effect of soil characteristics in severe forms of erosion, laboratory analyses were undertaken to determine the physico-chemical soil properties.

The temporal land use/cover analysis done using the post-classification change detection indicated that intact vegetation has undergone a significant decline from 1972 to 2006. The temporal changes within the intermediate years are characterized by cyclic transitions of decline and recovery of intact vegetation. An overall decline in intact vegetation cover, an increase in degraded vegetation and bare eroded soil was noted. Fragmentation analyses done in the communal villages of the central Keiskamma catchment indicated increasing vegetation fragmentation manifested by an increase in smaller and less connected vegetation patches, and a subsequent increase of bare and degraded soil patches which are much bigger and more connected. The Landscape Organisation Index revealed very low vegetation connectivity in the communal rangelands that have weak local traditional institutions. Fragmentation analyses in the riparian and proximal hillslopes revealed evidence of increasing vegetation fragmentation from 1972 to 2006. The Markov Cellular Automata simulation predicted a decline in intact vegetation and an increase in bare and degraded soil in 2019. The Keiskamma catchment was noted as experiencing high rates of soil loss that are above provincial and national averages. The classification of erosion features and valley infill showcased the vegetation enrichment in the ephemeral streams which is occurring at the expense of high soil losses from severe gully erosion on the hillslopes. This in turn has led to an inversion of grazing patterns within the catchment, such that grazing is now concentrated within the ephemeral stream channels. Soil chemical analyses revealed a high sodium content and low soluble salt concentration, which promote soil dispersion, piping and gully erosion. The presence of high amounts of illite-smectite in the catchment also accounts for the highly dispersive nature of the soil even at low SAR values. Significant amounts of swelling 2:1 silicate clays such as smectites cause cracking and contribute to the development of piping and gullying in the catchment.

Given the worsening degradation trends in the communal areas, a systematic re-allocation of state land in sections of the catchment that belonged to the former commercial farms is recommended to alleviate anthropogenic pressure. Strengthening local institutions that effectively monitor and manage natural resources will be required in order to maintain

optimum flow regimes in rivers and curb thicket degradation. Measures to curb environmental degradation in the Keiskamma catchment should encompass suitable ecological interventions that are sensitive to the socio-economic challenges facing the people in communal areas.

Acknowledgements

This research was made possible through postgraduate funding from NMMU Research Office and the NRF, I value this immense contribution. The outstanding support offered by several scholars added value to this PHD; indeed I was standing on the shoulders of giants. I am greatly indebted to my promoter Professor Vincent Kakembo for this guidance and support throughout my doctoral studies. I am forever grateful for his accessibility and guidance during my research. My sincere gratitude goes to Professor Kyoung Jae Lim the SATEEC software developer at Kangwon National University, Chuncheon, and Kangwon, Korea for reviewing my work on soil erosion risk modeling. I am also grateful to Dr Courage Kamusoko of Asia Air Survey, Kanagawa, Japan for his advice on the Markov Cellular model. I owe deep gratitude to Dr Pardon Muchaonyerwa for his assistance with soil analysis. Many thanks for making arrangements for soil analysis at Dohne Soil laboratory and for reviewing my soil analysis results. I also treasure the support offered by staff at the CSIR Satellite Application Centre in providing satellite imagery, expertise and image processing software.

I am greatly indebted to Dr John Odindi for his encouragement to purse this doctorate. He was willing to share his resources during difficult times. May I also extend my appreciation to the Department of Water Affairs and Forestry, South Africa and Mr. Peter Bradshaw of SANParks for providing critical digital datasets for this study. Finally, I would like to thank all my colleagues at NMMU particularly Adolf Nyamugama and Munyaradzi Manjoro for volunteering as field assistants during their own busy schedules. Many thanks go to staff in the NMMU Geosciences Department for creating an academically friendly atmosphere, which made this work worthwhile. I am much obliged to my entire family for their moral support. My deep appreciation goes to my wife Sophia and sons Tadiwanashe and Tawananyasha for their unwavering support during my studies.

To God be the Glory.

Dedication

I dedicate this to my parents who worked tirelessly to ensure I reach the apex of education.

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Chapter 1: Introduction

1.1 Introduction

Land degradation is a severe environmental problem confronting the world today (Al-Dousari *et al*., 2000; UNCED, 1992; Reynolds and Smith, 2002; UNEP, 2006). It has detrimental impacts on agricultural productivity and on ecological function that ultimately affect human sustenance and quality of life. It is estimated that 15% of the world's land has been destroyed by human induced soil degradation associated with soil erosion, salinization, nutrient loss and physical compaction. According to Badger *et al*. (2000), at least one billion people around the world are at risk of the impacts of land degradation while over 250 million are directly affected. The global financial costs associated with soil erosion are estimated around $26 billion per year, of which developing countries contribute $12 billion (UNEP, 1986). In Africa 494.2 million hectares (16%) of the land is degraded (Olderman *et al*., 1990; Oldeman 1994). A significant amount of land degradation related to injudicious land use has occurred in many parts of Africa. In Southern Africa, land degradation is recognized as a severe and prevalent environmental predicament (Snyman, 1998; Hoffman *et al*., 1999; Hoffman and Todd, 2000; Scholes and Biggs, 2004). South Africa acknowledges the gravity of land degradation, as a signatory of the UN Convention to Combat Desertification. Land degradation in South Africa is widespread, it is estimated that over 70% of South Africa has been affected by different types of soil erosion of varying intensities (Le Roux *et al*., 2007). The Eastern Cape Province is ranked as one of the most degraded provinces in South Africa alongside KwaZulu-Natal and Limpopo Province. The soil degradation index values are particularly high in communal areas due to a protracted history of environmental and political disregard dating back from the 1930's and accelerating in the 1960's (Ross, 1999). Land degradation in the Keiskamma catchment in the Eastern Cape Province, South Africa has reached alarming rates (Hill *et al*., 1977; D' Huyvetter, 1985; Rose *et al*., 1987; Marker, 1988; Rowntree and Dollar, 1994; Colloty, 1997; Matoti, 1999).

Land degradation can be explained as the long-term loss of ecosystem function and productivity induced by disturbances that make recovery unattainable without intervention (Bai *et al*., 2008). Muchena (2008) notes that land degradation happens gradually and cumulatively, and impacts negatively on vulnerable rural people dependent on the land. Land degradation is triggered by detrimental human impacts acting in complicity with extreme and persistent natural forces that stress ecosystems (WMO, 2005). The interaction of these processes determines the intensity of degradation. The agents of land degradation can be viewed as bio-geophysical and socioeconomic-political (Gad, 2008). Bio-geophysical causes include factors such as land use and land management, climate, biodiversity, terrain and soil type. Socioeconomic-political forces that influence degradation include land tenure, institutional support, income, human health, incentives and political stability. Olderman *et al*. (1990) highlight that overgrazing and agricultural activities are the main instigators of land degradation in Africa.

Soil degradation is a critical component of land degradation which comprises water and wind erosion, chemical degradation, excessive salts, physical degradation and biological degradation. Soil erosion is the most aggressive constituent of land degradation and has catastrophic impacts on fragile agricultural and natural ecosystems. The effects of soil erosion include on-site damage and destructive downstream or off-site consequences. Lal (2001) indicates that soil erosion is a manifestation of soil degradation that has implications for soil quality and productivity. Problems linked to soil erosion include loss of fertile topsoil for cropping, siltation, eutrophication, damage to infrastructure, loss of aquatic biodiversity and emission of carbon dioxide and methane gases that cause global warming (Nearing *et al*., 2004; Morgan, 2005; Onyando *et al*., 2005). Factors that interact to generate and cause soil erosion include soil vegetation/ground cover, soil erodibility, topography, rainfall erosivity and conservation practices. The predicament of soil erosion is exacerbated in poorly managed arid and semi-arid areas which are characterized by sparse vegetation cover, steep slopes and dominated by highly erodible soils. An assessment of the spatial distribution of soil erosion thus requires insights of how these parameters interact across different spatial and temporal scales (Le Roux *et al*., 2007).

Land use and land cover change significantly influence land degradation processes such as soil erosion and is an important input variable into global change research. Li *et al*. (2009) and Verburg *et al*. (2009) point out that land use/cover changes are the primary causes of soil degradation which directly impact ecosystem services that support human needs worldwide. Land use and land cover significantly affect key ecological functions and the critical issue of global environmental change (Dong *et al*., 2009; De Chazal and Rounsevell, 2009). Anthropogenic impacts significantly modify patterns of land cover change and increase landscape fragmentation (Muriuki *et al*., 2011). Dong *et al*. (2009) and Song *et al*. (2009) indicate that the spatial pattern of land use/cover is a manifestation of the underlying human and ecological processes. Riparian ecosystems for instance, are very sensitive to physical disturbances and environmental stress and among the most threatened ecosystems in the arid regions. Alterations of flow regimes have caused extensive ecological degradation and dearth of biodiversity (Kingsford, 2000; Jansson *et al*., 2000). Land use/cover change analyses and projection using remote sensing and Geographical Information System (GIS) provide a powerful tool to assess the temporal land cover trajectories and gain an understanding of key ecological processes and implications (Pelorosso *et al*., 2009). The analysis of the physical and chemical properties of the soils provides insights into the attendant erosion processes within sediment source areas. The present study seeks to assess land degradation within the Keiskamma catchment using temporal land use/cover change analysis, soil erosion modeling and soil physico-chemical analysis. Furthermore, this study intends to apply a land use cover change prediction model as a basis for informing policy formulation processes. The Keiskamma catchment is one of the catchments in the Eastern Cape, South Africa facing different forms of land degradation in the form of soil erosion, vegetation diminution and fragmentation. Keiskamma showcases the degradation going on in other impoverished communal areas of the Eastern Cape Province with a similar land tenure and land degradation history.

1.2 Problem Statement

Land degradation in most communal parts of the Keiskamma catchment has reached alarming proportions. The Keiskamma catchment is particularly predisposed to severe land degradation associated with soil erosion, thicket degradation and deteriorating riparian vegetation. There is a close coupling between land use/cover dynamics and degradation trends witnessed in the catchment. Soil erosion is prevalent in most of the communal areas in the catchment. Gullies are a major source of sediment in the catchment and are responsibility for a high proportion of soil loss. Hillslopes adjacent to communal areas are particularly vulnerable to gully erosion. The predominance of gully erosion in particular physiographic zones suggests that certain topographic zones are more susceptible to gully erosion that others. The soils are highly erodible and seem to play a major role in the development of piping and gully erosion in rangelands and abandoned fields. Riparian and hillslope zones are threatened by encroachment of alien vegetation, soil erosion and overgrazing. Impoundments in the form of damming upstream have negative effects on riparian and adjacent hillslope vegetation and increase degradation within these areas. The lack of an understanding of the trends in land use/cover change in the catchment currently impedes planning processes in the catchment. Analyses of the land cover changes occurring in sensitive ecological zones such as riparian zones are essential in assessing catchment health. There is also a growing need to quantitatively assess soil erosion risk potential at catchment scale using methods that integrate the major parameters that control soil erosion. On the basis of the gaps in knowledge outlined above, pertinent research questions to be tackled by this study are:

i) What are temporal land cover trends in the catchment?
ii) What are the future land cover scenarios in the catchment?
iii) What are the soil erosion and degradation patterns in the catchment?
iv) What are the topographic thresholds susceptible to gully erosion?
v) What is the relationship between soil properties and severe erosion forms?

1.3 Research Aim

The principal aim of this study is: To investigate land use/cover trends, model the spatial patterns of soil loss and predict future land use/cover scenarios.

1.4 Specific objectives

To achieve this fundamental aim, the following specific objectives are formulated.

1. To determine the magnitude and patterns of temporal land use/ cover change at catchment scale.

This objective is achieved by means of analyses of land use/cover change using post classification change detection. Object-oriented classification was performed using Landsat satellite imagery and Definiens Developer software. Specific landscape change detection analysis was done to highlight changes in the communal areas and the riparian and adjacent hillslope zones. Fragmentation analysis was performed by computing and analyzing landscape metrics in the riparian and adjacent hillslope areas to determine the land cover structural changes that have occurred since 1972. A similar landscape fragmentation analysis was done in the central part of the catchment dominated by communal settlements. Fieldwork was conducted to validate image classification results and to perform a Landscape Function Analysis, as well as appraisal of riparian zone condition.

2. To predict future land use patterns and land cover condition in the Keiskamma catchment.

The prediction was based on historical land use/cover trends observed in the catchment. Future land use/cover scenarios were simulated using as Markov-cellular automata model. Model validation was done to establish its feasibility in land use/cover projections in the catchment.

3. To determine the soil loss spatial patterns of the Keiskamma catchment.

This objective is achieved by using Sediment Assessment Tool for Effective Erosion Control (SATEEC), it is a GIS based Universal Soil Loss Equation (USLE) model that integrates sediment delivery ratios (SATEEC). Key parameters that were used in this model are rainfall erosivity, soil erodibility, land cover management factor, topography and conservation practice. Object oriented classification was used to map soil erosion surfaces and valley infill in ephemeral stream channels as a means of demonstrating the major sediment transfer processes operating in the Keiskamma catchment. Sediments are transferred mostly from rills and gullies (sediment sources) into ephemeral stream channels which act as sediment reservoirs.

4. To determine topographic zones susceptible to gully erosion.

The Mahalanobis distance method was applied in this study to compute the topographic thresholds and determine the susceptibility clusters to gully erosion. The gully locations were collected using a GPS in the field, additional points were acquired from SPOT 5 satellite imagery and aerial photography.

5. To characterize the physical and chemical properties of the soils and their implications to severe forms of erosion. A sub-objective was to establish whether significant differences exist in the sodic levels between the A and B soil horizons since the abandonment of cultivation in the 1950s and 60s.

This objective was achieved by conducting field surveys to observe gully development and piping. Soil samples were taken for laboratory analysis to determine the soil textural and chemical properties promoting piping and gully development. A student-test was used to establish the differences in the sodic concentrations between the A and B soil horizons.

6. To provide baseline recommendations for curbing land degradation and ensure future sustainability.

Having established land use/cover trends and soil erosion patterns in the catchment together with the drivers of land degradation, recommendations of possible methods to rehabilitate degraded areas and ensure future sustainability were made.

1.5 Chapter Outline

Chapter 1: Introduction

This chapter presents an overview of global land degradation problem and the Keiskamma catchment in particular. It also explores the common forms of land degradation such as soil erosion and land use/cover change. The main research problem is highlighted and the pertinent research questions are raised. The main aim of the research and its specific objectives are outlined. A synopsis of the methods used to accomplish each objective is given. An outline of the thesis chapters is provided.

Chapter 2: Literature view and study area

This chapter reviews literature on soil erosion modelling, soil dispersion and land cover classification and the geographical characterization of the Keiskamma catchment. It also examines the evolution of degradation in the Keiskamma catchment.

Chapter 3: An object based classification and fragmentation analyses of land use and cover change in the Keiskamma catchment.

The chapter analyses land use/cover changes and landscape fragmentation from 1972 to 2006 in the Keiskamma catchment. Temporal change detection was performed at catchment scale using object-oriented post-classification comparison. The chapter also determines the vegetation structural responses within the riparian and adjacent hillslope zones arising from impoundments of the Keiskamma River due to construction of Sandile and Binfield dams in 1983 and 1986 respectively. Further temporal change analysis involving the landscape fragmentation analysis in the riparian zones and the communal areas of central Keiskamma catchment was done by computing and analyzing landscape metrics. The chapter ends with a discussion and conclusion on the observed degradation trends and patterns.

Chapter 4: Projection LULC Trends in the Keiskamma using the Markovian Cellular Automata Analysis.

This chapter applies a stochastic model called Markov-cellular automata to project future land use/cover scenarios in the Keiskamma catchment. The prediction was based on historical land use/cover trends observed in the catchment. This analysis informs policy making, environmental planning and assessment protocols. The results of the Markovian modelling are discussed and the possible implications for the environment presented.

Chapter 5: Soil Erosion Risk Assessment of the Keiskamma Catchment using GIS and Remote Sensing.

This chapter examines the soil loss spatial patterns in the Keiskamma catchment using the GIS-based Sediment Assessment Tool for Effective Erosion Control (SATEEC) to assess the soil erosion risk of the catchment. SATEEC estimates soil loss and sediment yield within river catchments using the Revised Universal Soil Loss Equation (RUSLE) and a spatially distributed sediment delivery ratio. An exploration of empirical soil erosion models was done. The model calculates quantitative information on the processes of soil detachment and sediment deposition. Fieldwork was conducted to collect gully locations using a GPS. Topographic thresholds for gully erosion were determined using the Mahalanobis distance analysis. This chapter also characterizes the physical and chemical properties of the soils and their implications for severe forms of erosion. The differences in sodicity levels between the soil A and B horizons are investigated as a sub-objective using the student-t test. Object oriented classification was used to map to valley infill within ephemeral stream channels and erosion surfaces such as gullies. This phenomenon is prevalent in the semi-arid parts of the central Keiskamma catchment.

Chapter 6: Synthesis.

This chapter provides an integrative review of the results obtained in this study and their implications for the catchment health of the Keiskamma. The interaction of various parameters causing soil erosion in the catchment is explored, and general conclusions are drawn. The effectiveness of the techniques implemented in this study is also evaluated. A discussion of the possible interventions required to curb increasing degradation trends in the Keiskamma catchment is made. Future directions for research are proposed and the final conclusions of this study are drawn.

Chapter 2: A characterization of the Keiskamma catchment and review of land degradation assessment methods.

2.1 Introduction

This chapter is divided into two sections: a description of the study area and a review of the land degradation assessment methods.

2.2 Catchment setting

The Keiskamma is a semi-arid rural catchment located in the former Ciskei homeland of the Eastern Cape Province, South Africa (Figure 2.1). The catchment spans over 2 745 km² covering about 35% of the former Ciskei region (Hill *et al*., 1991) and has a population of about 223 000 people (DWAF, 2004). The Keiskamma is the main river in the catchment with headwaters in the Amatole Mountains which lie above Keiskammahoek town and flows eastwards for 263 km and drains into the Indian Ocean at Hamburg resort (33° 17´S 27° 29´E) (Colloty, 1997; Matoti, 1999). Its main tributaries are Tyume, Chalumna and Gulu.

The catchment is generally classified into three topographic zones namely: the escarpment zone, coastal plateau and the coastal zone (DWAF, 2004). The deeply incised Keiskamma River Valley bisects both the coastal plateau and coastal belt of the catchment. Isolated alluvial terraces and steep scarp zones characterize the incised valley. The coastal plateau covers most of the catchment at an altitude range of 600 to 900 meters above mean sea level and extends from the bottom of the Amatole mountain range. The coastal belt extends to a width of approximately 20km into the catchment. Between the coastal plateau and the catchment fringe is the escarpment zone referred to as the Hogsback. The distinguishing features of the escarpment are its steep slopes and high elevations of up to 1 938 meters above mean sea level and distinctively high rainfall. The Tyume is the main tributary of Keiskamma River, which has its headwaters in Hogsback escarpment.

2.2.1 Climate

Climatic variations in the catchment are highly correlated to elevation and proximity to the sea. The escarpment zone, which comprises mountain forests and pine plantations receives annual rainfall amounts of about 1 900mm while the semi arid coastal plateau receives 400-600mm, with most of rainfall received in summer. The mean annual rainfall is spatially distributed according to the topographic zonation of the catchment.

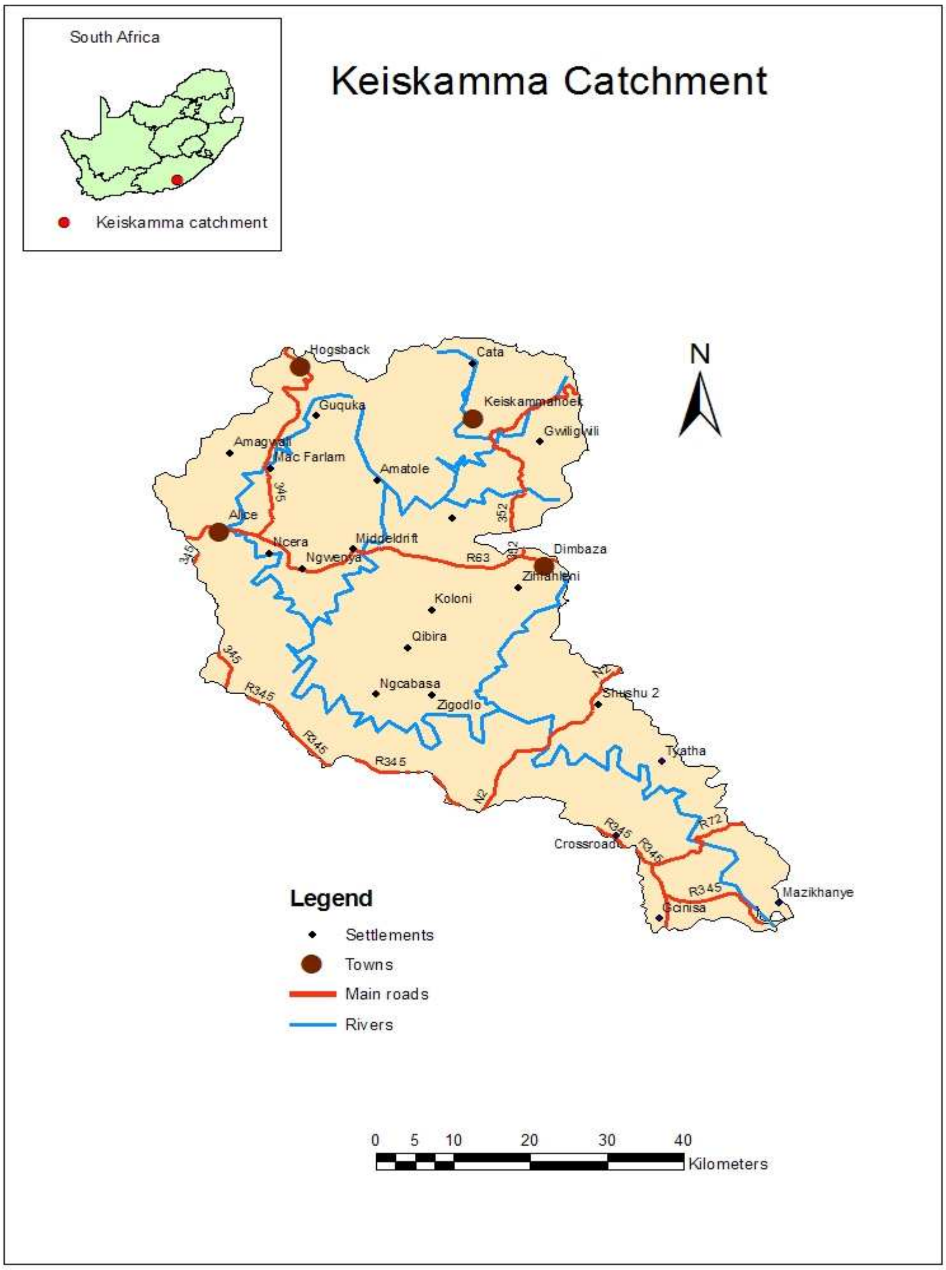


Figure 2.1 Keiskamma catchment

The summer months receive most rainfall while June and July winter months are the driest (Tanga, 1992). Large areas in the escarpment zone are protected and its land cover conditions can be described as pristine. Average annual temperatures are 11°C for the escarpment zone and 18°C for the coastal plateau. Temperatures can rise and fall to 38°C and -2°C in winter and summer respectively (DWAF, 2004). Whereas summer temperatures regularly exceed 40ºC, cold temperatures are experienced during winter months with occasional snowfalls in areas between the Amatole mountain range and Keiskammahoek to the Hogsback area. This contrasts with coastal areas where temperature variations are less pronounced.

2.2.2 Geological Setting

The geology of Keiskamma catchment is mainly underlain by the Beaufort series of the Karoo supergroup (Johnson and Keyser, 1976). The catchment is predominately characterised by sedimentary rocks such as shales, mudstones and sandstones (Weaver, 1991). The Beaufort series is generally comprised of a yielding sequence of sedimentary lithology grading from mudstone to sandstone (Story, 1952). The Karoo sequence consists of highly erodible shales, mudstones and sandstones of the Ecca and Beaufort series (D'Huyvetter, 1985). Witteberg sandstone/quartzite sinks beneath the Karoo system and is of biogeographical significance (Russell and Robinson, 1981). The catchments' coastal boundary is made up of unconsolidated beach sand and high coastal dunes and semi-consolidated sand overlying the older sedimentary rocks. Soil distribution in the Keiskamma catchment is highly correlated to the underlying geology and physiographic climatic patterns in the region (Verdoodt *et al.*, 2003). The recursive catena pattern of shallow rocky soils on the upper slopes and deep fine-textured soils in the valley floors is manifested in many river valleys through the Eastern Cape (Verdoodt *et al.*, 2003). Beaufort mudstones and minor dolerite intrusions characterize the geology of the escarpment zone (Johnson and Keyser, 1976). Highly erodible soils derived from shales and mudstones are prevalent throughout the catchment. Dominant soil forms include the Glenrosa, Misaphs, Oakleaf, Shortland, Hutton, Arcadia and Valrivier (Weaver, 1991). Stable and well structured soils derived from dolerite found in the catchment include Arcadia, Mayo, Milkwood and Shortlands (D'Huyvetter, 1985). The existence of swelling hydrous mica clays have also been identified by

Kakembo (2003). The topsoil is highly vulnerable to erosion due to the dispersive character of the soils inherited from the underlying geology and is further exacerbated by the removal of vegetation.

2.2.3 Vegetation

Three veld types identified in the catchment include Valley bushveld (Subtropical Transitional Thicket), Dohne sourveld and False Thornveld of the Eastern Cape (Acocks 1988). This classification was based on agricultural potential. A later vegetation classification based on phytosociological approaches was proposed by Louw and Rebelo (1996). This classification identified six vegetation types that consist of Coastal Forest and Afromontane Forest of the Forest biome, Valley Thicket of the Thicket biome, Sub-arid Thorn and Eastern Thorn Bushveld of the Savannah Biome (Lubke and Brenkamp 1996). The natural vegetation coverage from the coastal areas trending through to the escarpment include coastal grasslands and savanna (thornveld or sourveld).The river valleys are mostly characterized by dense bush of woody shrubs and trees known as the valley thicket (Louw and Rebelo, 1996). The high rainfall escarpment zone is covered by indigenous commercial forests. Palmer *et al.* (1988) attribute the variation in vegetation at smaller scale to the influence of climate and pedology. Alien plant invasions by *Acacia mearnsii* (black wattle); *Acacia dealbata (*Silver Wattle) and eucalypt species are noted throughout the catchment (DWAF, 2004). The upper Keiskamma and its Tyume tributary have higher densities of the invasive wattle. The growth of alien weeds such as *Lantana* and *Sesbani* among riparian vegetation has been observed (Amathole, 2009). The natural vegetation diversity has been degraded by alien plant invasions, overgrazing, and wood gathering, veld burning and poor farming activities. The semi-arid areas are characterized by patchy vegetation patterns. Degraded Valley Bushveld occupies most parts of the central Keiskamma area, while poor grass species are dominant in abandoned arable lands (Amathole, 2009).

2.2.4 Land use and land tenure

The Keiskamma catchment is largely communal and economically deprived (DWAF, 2004). Agricultural activities in the catchment include subsistence dry land and irrigated cultivation, and stock grazing. Irrigation schemes with total coverage of less than 1 500 hectares are located in the upper catchment which includes Keiskammahoek which covers 854 hectares, Zanyokwe 471 with hectares and Tyume with 231 hectares (DWAF, 2004). Commercial forestry covers less than 1 000 hectares and is located in the higher rainfall areas situated on the escarpment of Hogsback and the Upper catchment in the Amatola mountain range (DWAF, 2004). Scattered rural type villages typify the main form of residential settlement in the former Ciskei homeland. Five categories of land tenure which exist encompass the Tribal land, sometimes coupled with the quitrent system which constitutes over 80% of the former homelands areas, Freehold land, State land, Municipal land and Institutional land (DWAF, 2004). Communal land use patterns have largely been structured by the Native Land Act of 1913 which resulted in population explosions and environmental degradation in Bantu homelands (Moyo *et al*., 2008). Land use patterns found in the former Ciskei areas are a result of the betterment schemes of the 1960s whose objective was to curtail land degradation but are reported to have accelerated it (Kakembo, 2001; de Bruyn *et al*., 2000). The betterment programme shifted the more scattered rural settlements into more defined and nucleated villages with fences to demarcate cropping and grazing areas with reduced livestock (De Wet, 1987; De Wet, 1995; Bennett and Barrett, 2007). Trollope and Coetzee (1975) reveal that 78.7% of Ciskei was subject to the betterment programme. De-agrarianisation is rampant in Keiskamma, as borne out by the widespread land abandonment of cultivated land in the communal areas and former commercial farms owned by white farmers (Manona, 1999; Hebinck, 2007). There is increased reliance on remittance from urban wages and less dependence on cultivation (Manona, 1999).

2.2.5 Land degradation in the Keiskamma catchment

Many of the communal villages in the central part of Keikamma catchment are highly degraded and are particularly vulnerable to change. The villages are bordered by former commercial farms that were expropriated from the white farmers in 1981 and incorporated into the former Ciskei homeland (Kakembo and Rowntree, 2003). Land degradation in the Keiskamma catchment is caused by a number of physical and anthropogenic factors (Marker 1988, Rowntree *et al*., 2004). The physical factors that accelerate land degradation include soil of a sodic nature, erratic rainfall and steep slopes (Weaver, 1991; D'Huyvetter, 1985). Anthropogenic factors that drive land degradation in the catchment are overgrazing, cultivation, deforestation and alteration of river flow regimes through impoundments.

Overgrazing has been singled out as the major contributor of rangeland degradation in the catchment (Laker, 2000; Colloty, 1997, Bennett and Barrett , 2007; Bennet 2008, Kakembo and Rowntree, 2003; Moyo *et al*., 2008, Bennett *et al*., 2010). Bennett and Barrett (2007) studied the grazing management systems in the Keiskamma catchment and identified considerable differences in the management systems. They concluded that the degree of control the communities had on communal grazing resources had implications for degradation patterns observed in the catchment. The pressure exerted on grazing resources at a local level is influenced by the social and ecological heterogeneity that characterize the catchment (Bennett and Barrett, 2007). Rangeland degradation is exacerbated by the effects of weak local institutions which are unable to define and enforce rights to a particular grazing resource (Bennett, 2008). According to Moyo *et al*. (2008), local-level institutions and structures monitoring access and use of grazing lands are absent or ineffective in most villages. Other factors that cause conflict and lack of commitment in the management of grazing systems include the diversity of user groups, lack of grazing land as well as political and ethnic polarizations (Bennett *et al.*, 2010). Bennett (2008) indicate that many of these issues are a legacy of past apartheid polices such as forced resettlement and betterment planning. Open access grazing continues to threaten the rangeland resources in the communal areas of the Keiskamma catchment (Moyo *et al*., 2008). It is estimated that stocking density for the Keiskamma has been exceeded by 173% (Colloty, 1997). Although the maximum stocking density is about 45 000 livestock units, an assessment done in 1991 indicated that the stocking density was 78 000 livestock units (Colloty, 1997).

Fluvial systems are particularly sensitive to human impacts such as injudicious land use and other direct human impacts such as channel impoundments (Rowntree and Dollar, 2008). A study by Rowntree and Dollar (1994) highlights that the construction of the Sandile Dam in 1981 on the Keiskamma river had severe impacts on the natural function of the Keiskamma river such as amelioration of flooding, reduction in sediment transport capacity, channel siltation and general ecological problems (Rowntree and Dollar, 2008). The flow regimes in the Keiskamma river are highly regulated by impoundments and the dam has an estimated sediment trap efficiency of 100% (Rowntree and Dollar, 2008). The impoundments absorb most of the high discharges and reduce the natural flow regimes of the Keiskamma River (Rowntree and Dollar, 2008). Channel sedimentation and downstream aggradation have ever since increased due to the elimination of low-frequency high-magnitude flows responsible for the periodic flushing of the fluvial system (McGregor, 1999). Flood frequency curves show a reduction in peak flows of up to 30% since the impoundment in 1981 (Rowntree and Dollar 1994). The effectiveness of the impoundment was demonstrated in 1993 when a single rainstorm produced a discharge of 2.46 m^3s^{-1} in upstream part of the Keiskamma River and its tributary the Amatole River while a discharge of 0.1 m^3s^{-1} was recorded in the downstream parts of the Keiskamma main channel (Rowntree and Dollar, 2008). Reduction in flow regimes affects downstream hydrological and geomorphic processes (Stromberg *et al*., 1993; Stromberg, 1998). Channel siltation and tributary bars at channel confluences along the Keiskamma were observed by Rowntree and Dollar (1994).

The downstream hydrological impacts of river impoundments such as loss of saturation affect the plant water requirements for riparian vegetation and adjacent hillslope vegetation (Stromberg and Tiller, 1996; Stromberg *et al*., 1996; DeWine and Cooper, 2007; González *et al*., 2010). Downstream geomorphologic processes such as channel narrowing, sediment transportation and deposition and tributary bar formation affect the riparian vegetation condition and other ecological processes (Dominick and O'Neil, 1998; Friedman *et al.*, 1998; Grams and Schmidt, 2002). The long term effect of the reduction in flow regimes and changes in geomorphologic processes could be detrimental to vegetation species in the corridors adjacent the river (Nilsson *et al*., 1991; Nilsson and Berggren, 2000; Stromberg, 2001; Shafroth *et al*., 2002; Haney *et al*., 2008). Further degradation of riparian and adjacent hillslope vegetation is further excerbated by overgrazing and agricultural activity in these areas. Increased degradation in these sensitive ecological zones leads to the loss of biodiversity and deterioration of water quality due to sedimentation (Nilsson and Berggren, 2000; Johnson *et al*., 1995; Thomas, 1996; Shafroth *et al*., 2002; Haney *et al*., 2008; González *et al*., 2010).

Palmer (2010) indicates that blue bush and renosterbos (*Elytropappus rhinocerotis*) continue to invade disturbed and undisturbed rangeland in the Keiskamma river valley. Both species have been known to be invasive in the absence of fire and tend to favour areas which have suffered excessive soil disturbance (Kakembo *et al*., 2006, Kakembo *et al*., 2007, Palmer, 2010). Soil disturbances by cultivation, excessive trampling and subterranean activity by soil organisms such as termites and mole rats create niches favourable for the establishment of these invasive species (Kakembo, 2001; Palmer, 2010). The spatial variation of soil erosion in the former Ciskei catchments is related to its geology, soil and slope aspects (Weaver, 1991). Land degradation in the Keiskamma manifests itself in sheet erosion, rills, extensive gullying characterized by piping and subsurface erosion (tunnelling), loss of vegetation, bush encroachment, alien plant invasion, deteriorating riparian vegetation, bank erosion, siltation and poor water quality.

2.2.5.1 Soil erosion

Soil erosion is prevalent throughout most communal parts of Keiskamma (Hensley and Laker, 1975, 1978; D'Huyvetter, 1985; Kakembo, 2001). Causes of land degradation in these areas are linked to poor veld management, overgrazing, uncontrolled burning and deforestation associated with harvesting of firewood and medicinal plants. Injurious cultivation of highly erodible soils also triggered accelerated soil erosion (D'Huyvetter, 1985; Laker, 1978). Although land degradation is instigated by a complex interaction of physical, climatic and socio-economic variables, land tenure has been singled out as the most influential feature driving the intensity, rate and extent of degradation in South Africa (Meadows and Hoffman, 2002). Marker (1988) indicates that population explosions in the communal areas also contributed to increased soil erosion. Ephemeral stream channels have been observed in the field as the major sediment reservoirs in the semi arid parts of the catchment. The accumulation of sediment within ephemeral streams promotes vegetation within the stream channels. In summary this review indicates that soil erosion in the communal areas is driven by a multitude of factors that include land tenure, overstocking, imprudent land use planning, overpopulation, socio-economic variables, topography and unstable soils.

2.2.5.2 Sodic soils and severe erosion in the Keiskamma catchment

The excessive rates of soil erosion being experienced in the Keiskamma catchment are to some extent a result of soil structural problems evident in many parts of the study area such as surface crusting, hardsetting, slaking, swelling, and dispersion of clays. These soil conditions affect the water-holding capacity of the soil and promote high runoff and soil erosion. This ultimately increases sediment yields in the catchment. Extensive field observations in the most extensively eroded sites suggest that the soils in the catchment are sodic and highly dispersive. Surface crusting and sealing were also evident in most parts of the catchment.

A sequence of processes precedes the disintegration and erosion of sodic soils. These processes include slaking, dispersion, sealing, crusting, hardsetting and piping (Qadir and Schubert, 2002). Slaking arises as result of fragmentation of macroaggregates into microaggregates on wetting and leads to reduction in the number and size of pores on the soil surface, resulting in reduced water infiltration (Qadir and Schubert, 2002; Faulkner *et al*., 2003; Igwe, 2005; Qadir *et al*., 2006). Dispersion occurs when the repulsive electrical forces between individual clay particles surpass the attractive van de Waal's forces such that when the clay gets in contact with water, individual clay particles are progressively detached from the surface and get into suspension or washed away if the water is flowing (Qadir and Schubert, 2002; Igwe, 2005; Rhoton *et al*., 2007; Van Zijl, 2010; Verachtert *et al*., 2010). Dispersion is responsible for the high erodibility of soils and their susceptibility to piping (Faulkner *et al*., 2004; Jones, 2010). Spontaneous dispersion and freeing of clay particles from soil aggregates occurs when extensive hydration takes place (Qadir and Schubert, 2002). De Santis *et al*. (2010) indicate that soils which disperse spontaneously have severe problems with crusting and reduced porosity. Addition of electrolytes such as Ca^{+2} induces osmotic effects, resulting in the dehydration of the clay-water system, and minimizes the separation of clay particles in a process called flocculation (Rengasamy and Sumner, 1998). Oster and Jayawardane (1998) point out that exposed soil aggregated are at risk to degradation processes due to the stress induced by rapid water uptake, release of entrapped air, mechanical impact and stirring action arising from the flowing water. Low electrolyte concentration and high sodium and magnesium levels expose surface soils and make them more unstable in comparison to underlying soils (Qadir and Schubert, 2002; De Santis *et al*., 2010).

The effect of slaking and dispersion brings about reconfiguration of soil particles on drying, producing a densely packed thin soil with high shear strength in a process called structural crust or seal (Qadir and Schubert, 2002; Qadir *et al*., 2006; Summa *et al*., 2007; Ramenzanpour *et al*., 2010). Crust development is firstly linked to the physical break up of soil aggregates and their compaction. Secondly the dispersion and movement of clay particles into a region of 0.1- to 0.5mm depth where they settle and clog the conducting pores, thus reducing steady-state infiltration (Qadir and Schubert, 2002; Mills and Fey, 2003; Fox and Wilson, 2010). Whilst it is accepted that the two processes take place simultaneously, the physical breakup of soil masses accelerates dispersion and clay movement. Faulkner *et al*. (2004) and Igwe (2005) concur that the physical disintegration is determined by the type and concentration of cations in the soil solution and amount of water impacting on the soil. Other processes which reduce infiltration are hardsetting and sealing. Sealing affects the first 0.1 to 0.5mm below the surface and hardsetting displays massive, compact, and hard conditions in the entire ploughing zone of the upper soil layer (Bryan, 2000; Mills and Fey, 2003; Rhoton *et al*., 2007). Detrimental effects of hardsetting include the reduction of the infiltration rate, accelerated runoff and erosion (Bryan, 2000; Mills and Fey, 2003; Ramenzanpour *et al*., 2010). Other effects include impairment of water movement into the soil and reduce plant seedling emergence, subsequently affecting vegetation cover (Mills and Fey, 2003).

Piping is directly linked to the physical and chemical properties of the parent materials, particularly sodicity and dispersive nature (Rienks *et al*., 2000). Verachtert *et al*. (2010) described piping as subsurface erosion involving the removal of subsurface soils in pipe-like erosional channels to a free or escape exit. Pipe or tunnel erosion is a result of the concentrated subsurface flows through macropores, such as desiccation cracks or small rock fractures (Díaz *et al*., 2007; Richards and Reddy, 2007). It has long been established that dispersive soils contribute to pipe development. Studies reveal that the occurrence of piping is associated with particular physico-chemical and geomorphologic conditions (Faulkner *et al*., 2000; Faulkner *et al*., 2003; Díaz *et al*., 2007; De Santis *et al*., 2010; Verachtert *et al*., 2010). De Santis *et al*. (2010) and Verachtert *et al*. (2010) stress the significance of soil mineralogy in the emergence of piping erosion. Higher sodium to calcium plus magnesium ratios promotes the swelling and cracking potential of montmorillonitic soils (Faulkner *et al*., 2000; Faulkner *et al*., 2003; De Santis *et al*., 2010; Verachtert *et al*., 2010). Díaz *et al*. (2007) cited several field and laboratory experiments which confirmed the role of soil properties in piping erosion. High sodium ions deflocculate the clay fraction of soil aggregates and induce high susceptibility to erosion (Summa *et al*., 2007; Van Zijl, 2010). The presence of sodium causes dispersion of double layer clays and subsequently enlarges subsurface pipes (Faulkner *et al*. 2003; Jones, 2010; Verachtert *et al.*, 2010). Pipe roofs subsequently collapse to create deep gullies due to the loss of material strength as pipes get bigger (Rienks *et al*. 2000; Richards and Reddy, 2007). Pipe development is dominant is areas with dispersive sodic smectites with high shrinking and swelling capacity and low electrolyte concentration (Jones, 2010; De Santis *et al*., 2010). Piping is most common in clay, silt, fine sand, colluviums, alluvium, claystone, siltsone, mudstone, loess, tuff and volcanic ash and some soils (Rienks *et al*., 2000; Richards and Reddy, 2007; Jones, 2010; Verachtert *et al*. 2010). Jones (2010) pointed out that dispersion is in fact one of the major factors influencing piping and the erodibility of soils. The presence of piping or tunnel erosion exacerbating gully erosion in the Keiskamma catchment is indicative of the dispersive nature of the soil.

2.2.5.3 Distribution of dispersive soils in South Africa

Dispersive soils are predominantly found in arid and semi-arid climates in Southern Africa which receive rainfall amounts less than 850mm per year (Bell and Maud, 1994). The occurrence of sodic soils is strongly related to parent materials with elevated amounts of Na^+ releasing weatherable minerals (Rienks *et al*. 2000; Paige-Green, 2008). Geological settings mostly associated with dispersive soils in South Africa include sedimentary rocks from the Beaufort Group, the Ecca Group, the Molteno Formation and the Dwyka Formation, all belonging to the Karoo Sequence (Bell and Walker, 2000). Soils derived from the Sundays River, Cretaceous Enon, and Kirkwood Formations of the Uitenhage Group exhibit dispersive characteristics (Bell and Maud, 1994). Dispersive clays have also been identified in parent materials belonging to the Cape Supergroup, typically the Witteberg, Bokkeveld and Table Mountain Groups. Most granites in low lying areas and granodiorites particularly of the Swaziland Basement Complex are linked to soils of a dispersive nature (Bell and Walker, 2000). The location of dispersive soils on relatively steep hill slope promotes the rapid development of gully erosion (Botha *et al*., 1994). Other settings associated with dispersive soils are, slope wash colluviums, flood plain deposits and lake bed deposits and residual soils. Conditions favourable to the development of dispersive clays in Southern Africa can be generalized as follows (Elges, 1985; Bell and Walker, 2000; Rienks *et al*., 2000; Mzezewa *et al*., 2003).

1. Low relief locations with rainfall amounts less than 850mm and with SAR (sodium absorption ratio) values greater than two. Soils derived from granites are predisposed to the risk of high ESP values at low altitudes, especially under anaerobic circumstances where iron is mobilized as Fe^{2+}. Low lying regions associated with the above mentioned geological formations have dispersive soils.

2. Areas largely dominated by 2:1 clays with high ESP values such as illite and smectite particularly montmorillonite and vermiculite. This is typically the case in mudstone, sandstone and siltstones with SAR values greater than 2.

3. The formation of dispersive soils in other arid parts of Southern Africa is repressed by the presence of free salts regardless of the high SAR values. These soils are prone to dispersion once leaching of free salts commences. Dispersion is absent in soils formed under intense weathering conditions and dominated by kaolinite.

2.3 Assessment of land degradation

This section reviews remote sensing and GIS techniques used to assess land degradation. It further assesses some of the field techniques which can be used to rapidly validate landscape conditions.

2.3.1 The role of remote sensing in mapping land degradation

Remote sensing is a cost effective tool for monitoring land use/cover changes that provides consistent and repetitive measurements of territorial processes (Prenzel, 2004; Treitz and Rogan, 2004; Wessels *et al*., 2004). It plays a critical role in mapping the spatial distribution of degradation features and enables understanding the causes of land degradation (Shrestha, 2005; Shrestha *et al*., 2005). Land degradation features such as gully erosion, bare areas and degraded vegetation can be detected directly using satellite imagery (Wessels *et al*., 2004; Taruvinga, 2009). Land use/cover change has been considered as one of the most prominent aspects of land degradation (Ringrose *et al*., 1996; Foody, 2001; Wessels *et al*., 2004; Wessels *et al*., 2007). Many studies have successfully applied remote sensing and geographical information systems in examining land use/cover change and land degradation (van Lynden and Mantel, 2001; Petit *et al.*, 2001; Southworth *et al.*, 2002; Wessels *et al*., 2007; Garedew *et al*., 2009).

Many earth observation satellite sensors have been designed to acquire information on the earth's surface at spatial and spectral resolutions ideal for mapping land cover and land degradation processes (Rogan and Chen, 2004). Landsat satellite imagery for instance has proved to be an invaluable asset in land use/cover change detection studies because of its huge archive of data which starts from 1972 up to date (Franklin, 2001; Franklin and Mulder, 2002). SPOT satellite imagery also has a consistent and long history which dates back to 1986 (SPOT, 2002). Notwithstanding the higher spatial resolution for SPOT imagery, Landsat series satellite imagery has a larger swath width and higher spectral resolution than SPOT, which makes it more suitable for land use/cover change at catchment scale. Remote sensing and GIS software permit the classification of land use/cover patterns and are able to compute land use/cover transitions. Temporal analysis of satellite imagery facilitates the effective analysis of change trajectories linked to dynamics of change processes (Brown *et al*., 2000; Garedew *et al*., 2009; Tsegayea *et al*., 2010). Houet *et al*. (2010) states that the goal of temporal series is not limited to analysing historical land use/cover trends but to simulate future temporal evolution of the landscape. Temporal analysis can also be used to interpolate land use/cover distributions between observed dates (Hepinstall *et al*., 2008; Houet *et al*., 2010). The most commonly used change detection techniques include post-classification, vector change analysis, image differencing, image ratioing, image regression, principal components analysis (PCA), and change vector analysis (Mas, 1999; Civco *et al*., 2002; Lu *et al*., 2003; Verbessel *et al*., 2010; Wang and Xu., 2010; Yu *et al*., 2011). Coppin *et al*. (2004) and Jensen (2005) provide a comprehensive review of these algorithms.

In recent years, remote sensing has witnessed a paradigm shift in image classification (Castilla and Hay, 2008; Blascke 2009).The object oriented classification approach has gained currency over traditional pixel-based image classification (Blaschke, 2005; Castilla and Hay, 2008; Gamanya *et al*., 2009; Blascke, 2009). It is widely accepted that image segmentation procedures such as multiresolution segmentation are able to segment imagery into meaningful image objects (Burnett and Blaschke, 2003; Benz *et al*., 2004). Image objects have a much closer relation with real-world objects, which increases the value of the final land use/cover classification (Castilla and Hay, 2008). This aspect cannot be addressed by standard per-pixel classification methods. Other benefits for object-based analysis include the use of texture, shape, and topological relationships in the classification procedure (Benz *et al*., 2004). Duveiller *et al*. (2008) indicates that improved segmentation algorithms and object oriented methods are able to delineate and classify land use/cover efficiently. Object oriented approaches have been used successfully to delineate forest boundaries and produce large scale maps and quantitative information (Radoux and Defourny, 2007). Object oriented classification has successfully been used to map tree crowns and forest stands by Hay *et al*. (2005) and Bunting and Lucas (2006) in British Columbia and Queensland, Australia respectively. The advantages of object oriented classification out weigh per-pixel classification at very high spatial resolutions (Blaschke, 2009; Im *et al*., 2008). While this review provides significant potential for object based classification in change detection, this concept has not yet gained momentum in South Africa.

2.3.2 Land use/cover change models

Land use/cover change models are used to understand the dynamics of land use/cover change and their implications for land degradation. Verburg *et al*. (2004) indicate that land use change models support the analysis of the causes and consequences of land use change. Land use change models have been used effectively to demonstrate land use change and its relationship with the underlying driving mechanisms (Verburg, 2006; Verburg *et al*., 2009). Zhu *et al*. (2010) stressed the importance of integrating land use change models into policy matters that drive land use change. Land use change models are particularly important for hypothesis testing and evaluating potential future scenarios (Sang *et al*., 2010). One of the most attractive applications of land use/cover change modelling is in predicting future land cover trends. The Markov chains for instance, have proved to be invaluable for modelling land use/cover change at a variety of spatial scales (Wu *et al*., 2006; Kamusoko *et al*., 2009; Sang *et al*., 2010). Markov models are particularly useful in predicting possible future conditions under different scenarios and serve as a valuable contribution in formulating appropriate rural policies (McDonald and Urban, 2006). A detailed theoretical background of land-use change models is provided by Briassoulis (2000), while Verburg *et al*. (2004) provide a concise overview of the future directions of land use models.

2.3.3 Fragmentation analysis

Analyzing landscape patterns and its changes is an effective way of assessing the impacts of land cover change on ecological function and provides an important means of inferring spatial patterns in relation to land use processes (Lafortezza *et al.*, 2010*)*. Landscape patterns reflect the complex interaction of ecological and anthropogenic variables (Li *et al*., 2010; Wu *et al*., 2011). Anthropogenic activity has the potential to alter landscape structure and ecological function of landscape over time (Alberti, 2010; Morris, 2010; Long *et al*., 2010). Landscape fragmentation is the process by which a landscape matrix is increasingly broken up into smaller and more isolated patches (Lafortezza *et al*., 2010; Ouyang *et al*., 2010). Measuring landscape fragmentation enables scholars to infer the impacts of human activity on ecological systems (Teixido *et al*., 2010). This is achieved by computing a variety of landscape metrics that reveal landscape fragmentation (Southworth *et al*., 2004; Long *et al*., 2010). Landscape metrics evaluate the spatial structure of land cover based on number, size, shape and configuration of patches of different land use/cover classes (Coops *et al*., 2010). These landscape metrics are used in conjunction with areal statistics to describe landscape structure and composition (Cerezo *et al*., 2010). Li and Reynolds (1994) indicate that landscape patterns are characterized by five aspects: (1) number of patch types; (2)

proportion of each patch type; (3) spatial arrangement of patches, namely patch aggregation level; (4) patch shape; and (5) contrast between neighbouring patches. These five components are therefore critical in determining the value of landscape metrics in quantifying spatial patterns. McGarigal *et al*. (2002) proposed a number of landscape metrics that can be used to monitor landscape fragmentation. The selection of landscape metrics is highly dependent on the application (Gergel 2007; Peng *et al*., 2007; Peng *et al*. 2010). The most widely used landscape metrics are: largest patch index, number of patches, patch density, mean patch size, landscape shape, index and interspersion and juxtaposition index. McGarigal *et al*. (2002) define these landscape metrics as:

(1) Number of patches (NP): the total number of patches in a class

(2) Largest patch index (LPI): area of the biggest patch in each class, expressed as a percentage to the total landscape area.

(3) Edge density (ED): sum of length of all edge segments for the class, divided by total landscape area.

(4) Patch density (PD): the NP per unit area.

(5) Mean patch size (MPS): the total area occupied by a specific patch class divided by the NP of that class.

(6) Landscape shape index (LSI): measures the amount of edge present in a landscape relative to what would be present in a landscape of the same size but with a simple geometric shape and no internal edge. It indicates the complexity of patch shape for a class (where an index of 1 represents a square, the index increases without limit as the patch becomes more irregular).

(7) Interspersion and juxtaposition index (IJI): indicates the degree of interspersion of a focal patch class with all other classes. When the class is found adjacent to only one other class type the IJI index is 0, this increases to 100 as the patch type becomes increasingly interspersed with other class types.

The integration of landscape fragmentation analyses within remote sensing provides a powerful approach to analyzing and describing spatio-temporal changes driven by anthropogenic impacts (Wu *et al*., 2000; Wegandra *et al*., 2004; Kamusoko and Aniya, 2007). While the integration of landscape metrics and remote sensing is highly valuable, the classification algorithms often affect the thematic resolution of the classified maps which subsequently affect the landscape metrics results (Baldwin *et al*., 2004;

Buyantuyev and Wu, 2007; Turner, 2005). This problem is particularly evident in most pixel-based classifications were the mixed pixel effect is common and results are often dependent on input values (Kelly *et al*., 2004; Kelly *et al*., 2011). The problem can be solved by using object-based image analysis where images are segmented into distinct patches, or image objects, based on spatial and spectral similarity before classification (Blaschke, 2010; Lang *et al*., 2008; Langanke *et al*., 2007; Kelly *et al*. 2011). Object oriented classification eliminates speckle and reduces the need for post-classification processing and smoothing filters (Kelly *et al.*, 2011). This new technique is increasingly proving to be more effective than per-pixel classifications in applications related to landscape structural analysis (Shiba and Itaya, 2006; Hay and Castilla, 2008; Jobin *et al*., 2008; Schöpfer *et al*., 2008; Blaschke, 2010).

2.3.4 Landscape function analysis

Landscape change is the alteration in the structure and function of the ecological mosaic over time (Walz, 2008). In recent years, a technique called landscape function analysis (LFA) has proved useful in quickly assessing landscape condition. This method is centred on the concept of landscape function coined by Ludwig and Tongway (1997) for Australian rangelands. LFA has now been adopted internationally for measuring and monitoring landscape function. This monitoring procedure uses rapidly acquired visual field indicators to assess the functional status of rangelands. The field indicators used for LFA reflect the status of key ecosystem processes (Whitford, 2002). Simple and easily measured attributes have been shown to indicate the functionality of a landscape (Tongway and Ludwig, 1997). A fundamental first step of LFA is assessment of landscape organisation, this method uses patch and fetch indicators to characterise the spatial pattern of resource loss or accumulation. Landscape organisation data are collected on a line transect oriented in the direction of resource flow (Tongway and Hindley, 2004b). Landscapes are considered to be highly functional if they effectively trap, store, concentrate, and utilize resources such as water and nutrients (Tongway, 2010). In contrast landscapes that lose resources to runoff and winds are considered dysfunctional. LFA facilitates rapid assessment of crucial processes linked to land degradation such as runoff, erosion, infiltration, plant growth and nutrient cycling. Functional (resource conserving) landscapes are considered to be in good range condition, while dysfunctional landscapes are regarded to be in poor condition (Tongway and Hindley, 2004a). Rezaei *et al*. (2006) used LFA to assess rangeland capability in Iran. Their study examined the functioning of the soil-landscape system and its effects on plant growth. Palmer *et al*. (2001) successfully used landscape organisation indices to establish differences between contrasting rangelands in Peddie district, South Africa.

2.3.5 Modelling soil erosion

Soil erosion modelling involves the processes of mathematically expressing soil particle detachment, transport, and deposition on land surfaces (Nearing, 1994). The development of mathematical models for estimating soil loss dates back to the 1940s. Zingg (1940) related soil loss to slope length and gradient and Smith (1941) included the influence of crops and conservation practices on soil loss. Furthermore, Musgrave (1947) formulated the Musgrave equation by adding the rainfall factor. Nearing *et al.* (1994) indicate that modelling soil erosion is done for at least three main reasons:

1. Predictive assessment of soils loss for soil erosion inventories and conservation planning.
2. Predicting the spatial and temporal occurrence of soil erosion using physically-based mathematical models. This is useful in targeting erosion hotspots for rehabilitation.
3. Enhancing the understanding of erosion processes and their interactions with each other.

Many models of soil erosion loss estimation have been developed (Wischmeier and Smith, 1978; Nearing *et al*., 1989; Renard *et al*., 1991; Adinarayana *et al*., 1999; D'Ambrosio *et al*., 2001; Veihe *et al*., 2001; Shen *et al*., 2003; Arnold *et al*., 1998; Morgan *et al*., 1998 Flanagan and Nearing, 1995). Soil erosion models are classified into three types namely: empirical, conceptual and physically models (Nearing *et al*., 1994). Empirical models are based on observations and inductive logic and are generally statistical in nature. The parameters for empirical models need to be calibrated and examples include the Universal Soil Loss Equation (USLE). The USLE method computes annual upland soil loss resulting from sheet and rill erosion in tonnes per hectare per year. The USLE model has proved to be the most practical method of estimating soil erosion potential for nearly 40 years (Dennis and Rorke, 1999; Kinnell, 2000). Other examples of empirical models are the Revised Universal Soil Loss Equation (RUSLE) and the Modified Universal Soil Loss Equation (MUSLE) which are improvements of the USLE model. The RUSLE computes annual soil loss and MUSLE proposed in 1972 computes sediment yield for a single storm event (Johnson *et al*., 2000). Empirical models are largely used in predicting soil loss and sediment yield estimates. Physically-based models are a synthesis of the individual components and mechanisms controlling soil erosion. They take account of complex interactions between several factors and their spatial and temporal variability. Physically-based models are useful in the identification of the most critical aspects of the soil erosion process where research and control efforts should be directed (Nearing *et al*., 1994). The applications of process-based models are however limited due to their intensive data and computation requirements. Conceptual models are an intermediate between physically-based and empirical models, which are based on spatially lumped forms of water and sediment continuity equations (Nearing *et al*., 1994). Conceptual models are often based on unit hydrograph theory and are used to predict sediment yields (Andrews *et al*., 2010; Bhunya *et al*., 2010).

Topography has a pervasive effect in hydrological processes and gully development in particular (Moore *et al*., 1991). Ancillary topographical derivates from a Digital Elevation Model (DEM) are useful in predicting potential ephemeral gully areas and can be used in tandem with satellite imagery to identify gullied areas (Kakembo *et al*., 2009). Topographical indices have significant potential in predicting hillslope areas susceptible to gullying (Moore *et al*., 1991; Thorne *et al*., 1986). Topographically Similar Areas (TSAs) tend to have similar hydrological and sediment source/storage attributes. Areas with similar topographical attributes such as gullies could be deemed as potential sediment sources. These areas could be identified using a topographic indices derived from a DEM and be regarded as vulnerable areas (Desmet and Govers, 1997). Several studies reveal that ephemeral gullies could be instigated by convergence of subsurface flow leading to saturation excess overland flow and saturation return flow (Moore and Burch, 1986; O’Loughlin, 1986; Thorne *et al*., 1986; Moore *et al*., 1988b; Poesen, 1993; Poesen *et al*., 1997; Souchére *et al*., 2003). Subsurface flow affects gully erosion through seepage flow and preferential flow through soil- pipes (Souchére *et al*., 2003). Topographic convergence and subsurface process are evidently an important mechanism which influence soil loss, but are often neglected in most soil erosion models (Moore *et al*., 1988; Baade *et al*., 1993; Baade, 1994; Huang and Laflen, 1996; Poesen *et al*., 1996). Mahalanobis distance method provides a valuable means to compute how similar a set of landscape conditions are to an ideal set of conditions (Farber and Radmon, 2002; Jenness, 2003). This method is ideal for discerning typical areas susceptible to gulling and piping takes into account the covariance among variables (Fernández *et al*., 2008). Topographic position, slope, aspect, surface curvature, profile, planform, upstream distance and contributing area are critical topographic variables which impact on gully formation since they influence the direction and magnitude of water flow (Moore *et al.*, 1988; Desmet and Govers, 1997; Desmet *et al*., 1999; Vandekerckhove *et al*., 1998; Vandekerckhove *et al*., 2000; Kakembo *et al*., 2009).

2.3.6 Assessment criteria for soil sodicity and dispersivity

The assessment of the chemical and physical properties of the soils is essential in understanding the intrinsic properties of the parent material at play in erosional dynamics. Field evidence of the underlying factors influencing soil sodicity and dispersion is critical in understanding the degradation trends.

Excessive accumulation of sodium (Na^+) in soils is problematic, as it alters the exchangeable and soil solution ions, soil pH and destabilizes the soil structure. The soil hydraulic properties are adversely affected, whilst soil susceptibility to crusting, runoff and erosion are significantly increased (Qadir and Schubert, 2002). The ensuing low electrolyte concentration in the soil solution promotes the adverse effects of the exchangeable (Na^+), such that dispersion occurs even at exchangeable sodium percentage (ESP) values of less than 5% (Qadir and Schubert, 2002). According to De Santis *et al.* (2010), the influence of the composition and concentrations of salts in the soil solution are conditioned by the amount of clay and the mineralogy of the clay fraction. They conclude that dispersion is detected by the chemical and mineralogical composition plus the free salt contents of the soils. Rahimi *et al*. (2010) pointed out that dispersion may occur in any soil with high exchange sodium percentage values, even in sand.

Bell and Maud (1994) suggested that the threshold for soil dispersion is SAR value greater than 2. This value is considered suitable for South Africa as attested by the study in Kwazulu Natal by Bell and Walker (2000) and Rienks *et al*. (2000). Beckedahl (1996) observed advanced piping and gullying in the former Transkei located in the Eastern Cape in materials with ESP values ranging from 0 to 19 and SAR values ranging from 0.3 to 21.9. A similar study by D'Huyvetter (1985) in Ciskei also confirms the threshold. The South African thresholds for soil dispersion are generally in line with the Australian definition due to similar environmental conditions (Sumner, 1993). An ESP of 6% is regarded as the critical limit for the undesirable effects of sodicity in Australia after a review by Northcote and Skene (1972). An ESP of above 15% is regarded by Gerber and Harmse (1987) as signifying a highly dispersive soil, while a value below 6 denotes non-dispersive soils. Bell and Walker (2000) revealed that soils become chemically dispersive as soon as they exceeded as an ESP value of 2.0%. in the Keiskamma catchment, severe forms of erosion were noted in soil derived from Beaufort mudstones and shales with ESP values as low as 2.5% (D'Huyvetter, 1985). Dispersion increases in clays dominated by illite when the Mg:Ca ratio is higher than Ca: Mg ratio (Yilmaz *et al*., 2005; Qadir *et al*., 2007). Lado and Ben-Hur (2004) reveals that illite suspensions tend to disperse more than montmorillonite suspensions under similar ESP and electrolyte concentrations. Illite clays are formed pedogenically by the conversion of expanding clays to less expanding minerals in arid and semi-arid environments (Fonseca *et al*., 2007; Caner et al., 2010). Rengasamy *et al*. (1984) note that red-brown soils dominant in illite are more susceptible to dispersion even at low sodium adsorption ratio and under weak mechanical forces. Several studies have found that illite had greater clay dispersion and clay movement compared to montmorillonites (Yilmaz *et al*., 2005; Calero *et al*., 2008; Fonseca *et al*., 2009; Igwe *et al*., 2009). An earlier study by (Goldberg *et al*., 1991) found that illite plays a critical function in influencing flocculation-dispersion processes in the interaction of solution pH and SAR value. The stability of the different clay minerals was ranked according to descending stability thus: hematite> kaolinite> chlorite> smectite> illite (Lado and Ben-Hur, 2004; Yilmaz et al. 2005; Igwe et al. 2009)

Research on soil dispersion in South Africa and elsewhere has largely remained inconclusive in terms of the thresholds of soil sodicity, as Sumner (1993) concludes that "no single simple definition is possible". Gerber and Harmse (1987) suggested a value of SAR higher than 10 as symptomatic of a dispersive soil. Mikailsoy and Pachepsky (2010) recommend a threshold value of ESP of 10% as characteristic of dispersion in soils that have their free salts leached by seepage of relatively pure water. There is no agreement in the classification of salt-affected soils and various schemes are used in different countries. Whereas the United States and Europe consider an ESP higher than 15% as critical limit for sodic soils, Australia considers an ESP of 6% as the threshold (Rengasamy *et al*., 1984). This divergence is attributed to variations in soil factors in the different localities such as electrolyte concentration, pH, organic matter and clay mineralogy affecting clay dispersion and subsequently the physical properties of the soil (Rengasamy and Sumner, 1998). SAR and Electrical conductivity (EC) of the soil solution largely influence clay dispersion when clay mineralogy, pH and organic matter are relatively constant (Rengasamy *et al*., 1984). Electrical conductivity is a proxy measure of soil salinity and not directly related to soil sodicity (Bell and Walker, 2000).

Several studies show evidence that there is no single threshold at which Na^+ initiates the degradation processes in soil (Sumner, 1998; Levy *et al*., 1998; Oster *et al*., 1999; Bell and Walker, 2000). Most countries have followed the definition of sodic soils suggested by the US Salinity Laboratory due to the simplicity of the numerical criteria. The US Salinity Laboratory (1954) proposed that an EC of 4 dSm^{-1} is the threshold of salinity, with values greater than 4 dSm^{-1} considered as saline. An ESP of 15% was marked as the threshold of sodicity, with higher values being sodic. They further recommended that saturated soil paste pH of 8.5 as the critical limit for non-saline-alkali (sodic) soils. Soils with ESP and pH values higher than the critical limits are regarded as being dispersive and encounter serious physical problems when wetted. Another category of saline-alkali (saline-sodic) was considered and fulfilled the criteria of EC $\geq$ 4 dSm^{-1} and ESP $\geq$ 15 with variable pH values commonly less than 8.5. This criterion for classifying sodic and saline soils has been challenged and rejected by many a scholar (Qadir and Schubert, 2002). Sumner *et al*. (1998) reviewed data from several studies and concluded that degradation processes may happen is some soils at ESP values much lower than the ones proposed by the US Salinity Laboratory Staff. The salinity threshold of 4 dSm^{-1} suggested by the US Salinity Laboratory (1954) is largely unrelated to the effects of salinity on the soil physical properties but to the effects of salinity on crops. Sumner *et al*. (1998) demonstrated that sodic soil behaviour may occur at ESP less than 5 even if the EC is lower than 4 dSm^{-1}.

Sumner *et al*. (1998) proposed a more appropriate classification criterion for sodic soils, which distinguishes soil dispersibility in three classes based on EC and SAR of 1:5 water extracts. The soil classes are spontaneously dispersive, mechanically dispersive and flocculated. Non-sodic ($SAR_{1.5} \leq 3$, $ESP \leq 6$), sodic ($SAR_{1.5} = 3\text{-}10$, $ESP = 6\text{-}15$) and very sodic ($SAR_{1.5} > 10$. $ESP > 15$). Soil salinity is also differentiated into three categories of salinity based on continuously variable $EC_{1.5}$. The salinity classes are non-saline, saline and very saline. The three sodic classes are combined with three salinity classes yielding a total of nine classes that account for the several factors that control soil sodicity (Qadir and Schubert, 2002).

Soil dispersion is also dependent on the cation exchange capacity (CEC). The cation exchange capacity is indicative of the type and nature of clay minerals present (Bell and Walker, 2000). Dispersive clays are related to 2:1 phyllosilicates with CEC values ranging from 40 to 150 meq/100g clay. Gerber and Harmse (1987) confirmed that non dispersive soils with an ESP of less than 6% had low cation exchange values (15 meq/100g of clay.). The ESP versus CEC chart developed by Gerber and Harmse (1987) is considered as one the most reliable chemical methods of determining dispersive soils (Bell and Walker, 2000). Rengasamy and Olsson (1991) proposed a classification of sodic soils based on SAR, electrolyte concentration and pH. In South Africa a SAR threshold of 2 and ESP of 5 are used to define dispersive soils (Bell and Walker, 2000; Paige-Green, 2008).

2.4 Conclusion

Although many scholars have studied various aspects of land degradation in the communal areas, the temporal land use/cover trajectories and landscape fragmentation trends have received little attention. Little is known about the long term response of downstream riparian and hillslope vegetation to physical disturbances which reduce the flow regimes. Furthermore, no studies have been undertaken to predict future land use/cover scenarios in Keiskamma catchment and many parts of the Eastern Cape. The role of soil physico-chemical characteristics in the development of pipes and gullies has also received little attention in previous studies. In addition, the spatial distribution of soil erosion at catchment scale and identification of active sites of erosion and deposition still need to be mapped accurately. This gap can be filled by applying remote sensing and GIS to model the spatio-temporal land use/cover dynamics occurring in the catchment as well as quantifying the spatial extent of soil erosion. The integrated application of remote sensing, GIS and mathematical modelling is a powerful means to measure land degradation processes over time. GIS is able to integrate the complex variables that affect soil erosion. Remote sensing enables one to do temporal analyses of land use/cover dynamics. Simulation models such as the Markov Cellular Automata can be used successfully to predict future land use/cover trends. Soil erosion models integrated within GIS software also provide an effective means to predict soil erosion processes. Remote sensing and GIS work however requires validation and calibration in the field. It is also evident from this literature review that particular processes such as gully erosion are topographically controlled. It is imperative that topographic thresholds of areas susceptible to gully erosion be derived to understand processes underlying the development of severe erosion forms.

Chapter 3: An object based classification and fragmentation analysis of land use and cover change in the Keiskamma catchment, Eastern Cape, South Africa

3.1 Introduction

Change detection of land use and land cover is a critical consideration for environmental health assessment. The inventory and monitoring of land-use/land-cover changes are indispensable aspects for the understanding of change mechanisms and modelling the variables at different scales (Turner *et al*., 1995; William *et al*., 1994). Remote sensing has shown significant potential in measuring the changes occurring on the earth's surface. Many studies have successfully applied remote sensing and geographical information systems in ecological studies (Turner *et al.,* 2003; Li *et al*., 2004; Kamusoko and Aniya, 2007; Giordano and Marini, 2008). Earth observation remote sensing is an effective tool in monitoring the spatio-temporal dynamics in landscape ecology. Change detection is a fundamental remote sensing technique which seeks to determine the environmental temporal changes from satellite and airborne sensed images. The difference in reflectance values between images acquired at two different times as a result of the physical changes on the land is the basis for all change detection techniques.

The advent of object-oriented segmentation and classification is a major paradigm shift from the traditional per-pixel classification. A fundamental problem for incorporating digital imagery into classification processes is that for a given land cover unit, spectral response is represented in digital imagery as series of discrete pixels covering a wide range of spectral values, yet for classification purposes, the land use unit is seen as a single homogeneous polygon (Hall, 2003). One solution to this dilemma is to aggregate the individual pixels representing the land use into an image object represented spectrally as the combined response of all underlying pixels. The image objects, rather than the underlying pixels, become the carriers of image information and form the basic units of the subsequent analysis (Chubey *et al*., 2006).

O'Neill (1997) points out that ecosystem health can be monitored effectively if spatial metrics which measure dominance, contagion and fractal dimension are monitored through time. These three landscape and structure metrics are incorporated in object-based techniques (Jensen, 2005). Multiresolution segmentation done at a suitable scale results in images objects that could be regarded as patches or ecological units that are suitable for further ecological analysis particularly landscape fragmentation analysis. Landscape fragmentation is defined as processes in which large continuous cover is subdivided into a number of smaller patches of smaller total area that are isolated from each other by a matrix of habitats unlike the original (FAO, 2007). Some of the effects of fragmentation on landscape structure are: decrease in the overall amount of habitat and mean patch size, increment of the edges, decrease of the core area and isolation of the habitat patches (Herold *et al*., 2003; FAO, 2007; Turan, 2010). Turner *et al*. (2001) suggest that a close relationship exists between landscape pattern and processes occurring on the landscape. Analyzing landscape transformations and structure in terms of composition and configuration is essential when evaluating its state and response to disturbances (O'Neil *et al*., 1999). Landscape metrics are used to quantify specific spatial characteristics of patches, classes of patches, or entire landscape mosaics. Kamusoko and Aniya (2007) indicate that analysis of land use/cover change and landscape structure is useful in understanding the extent and implications of fragmentation within landscapes.

Whereas many change detection studies have been done in South Africa using per-pixel methodologies, the application of object oriented techniques which integrate landscape approaches such as landscape fragmentation analysis has not attracted much attention. Object oriented classification provides a viable change detection method which is more suitable for landscape fragmentation than the traditional per-pixel classification. This is because object oriented classification generates homogeneous objects which closely resemble ecological patches. Indeed, the concepts encapsulated in object oriented classification are aligned with landscape pattern analysis (Shao and Wu, 2008).

Riparian zones in particular have a central ecological and economic role in most landscapes and are sensitive indicators of environmental change. They occupy the ecotone that acts as a buffer between aquatic and terrestrial systems (Nilsson *et al*., 1997; Nilsson and Berggren, 2000). The degradation of the riparian zone is driven by anthropogenic impacts such as deforestation, overgrazing and river impoundments. River impoundments in particular alter the water flow regime and impacts on downstream vegetation condition (Nichols *et al*., 2006). Vegetation conditions in the riparian and slope riverine proximal wetlands are a reflection of the constraints imposed by environmental conditions such as climatic, hydrologic regime and geomorphologic processes (Nilsson and Berggren, 2000). The riparian and slope riverine proximal wetlands are hydrologically and topographically adjoining and have a surface and shallow subsurface hydrologic connection with the river (Lee *et al*., 2008). The interactions between the river channel and riparian and adjacent hillslope vegetation generate and maintain important hydrologic and ecological functions (Lee *et al*., 2008). River impoundments affect riparian vegetation upstream by rising water levels. Downstream environments are affected by a decrease in the flow regimes which alters hydrologic processes such as riparian and hillslope saturation. Flooding regimes are also altered in terms of the timing and duration, which affect processes such as erosion, sedimentation and ground water recharge (Nilsson and Berggren, 2000). Permanent loss of saturation has severe implications for ecological function, as reflected by deteriorating vegetation condition. Such vegetation condition could be manifested by an increase in fragmentation such as shape of patches, decrease in patch size, habitat interspersion and connectivity and proportion of each land cover type on the landscape.

In the present study, land use/cover changes and landscape fragmentation from 1972 to 2006 are analysed in the Keiskamma catchment, Eastern Cape, South Africa. Temporal change detection was performed at catchment scale using object oriented post-classification comparison. A more in-depth temporal change investigation involving landscape fragmentation analysis in the riparian-hillslope proximal zones of the Keiskamma river and communal areas of central Keiskamma catchment was done by computing and analyzing landscape metrics.

3.2 Methodology

3.2.1 Remote sensing and GIS data

A combination of datasets comprising Landsat imagery, orthophotographs, a 20 m DEM and 1:50 000 topographic maps were used for the study.The Landsat series of satellites has the most comprehensive archive of earth observation satellite imagery to date and provides an excellent baseline resource for moderate resolution land cover change detection studies from 1972 (Chander *et al*., 2004; Chander *et al*., 2007). A total of six Landsat MSS and Landsat 5 TM images were selected for the study. The dates of acquisition for the images are 21 November 1972 for Landsat 1 MSS, 30 April 1988 for Land 4 MSS, 08 December 1993, 19 December 1997, 12 January 2001 and 12 December 2006 for Landsat 5 TM images. A deliberate effort was made to ensure all the images were acquired in the summer rainfall season during which the spectral differences between vegetated, cropped and degraded areas are distinct. The unavailability of images and cloud cover problems constrained selection of images of the same month. The Object-oriented post- classification method was used to classify the imagery. Historical datasets such aerial photographs, SPOT imagery and topographic maps were also acquired for accuracy assessment.

3.2.2 Image Pre-processing

Geometric and radiometric accuracy are a prerequisite for reliable change detection using satellite imagery. The orthorectification process corrects different viewing angles typical of multi-temporal datasets and also ensures that images and secondary products overlay perfectly with other GIS datasets. Toutin's Low Resolution satellite orbital model was used for orthorectification. The multi-temporal image datasets were georeferenced to an orthorectified 5m Spot mosaic and projected to the Universal Transverse Mercator (UTM) system using the World Geodetic System 1984 datum. A 20 m DEM was used to correct for relief displacement caused by local topography. At least 25 evenly distributed ground control points were selected for each image. The cubic convolution resampling technique was used and a root-mean-square error (RMSE) of less than 0.35 pixels was obtained for all the six images. Cubic convolution was used for resampling because it out performs simple nearest neighbour resampling in terms of geometric accuracy (Campbell, 2002). Geometric or positional accuracy is a critical factor in change detection studies. Although cubic convolution alters pixels values, this effect is not significant since the images are classified independently using object oriented classification which merges pixels into image object. The change detection comparison done in this study uses a thematic comparison of the land use/cover classes rather than pixel to pixel comparison. A semi-empirical quick atmospheric correction (QUAC) method available in ENVI 4.7 software was applied in this study to normalize the Landsat 5 TM and Landsat 1, 4 MSS imagery. The radiometric gains and bias of Landsat 5 TM determined at launch and available in the header files are now invalid due to the deteriorating radiometry of the aging Landsat 5 TM sensors. QUAC provides a viable alternative to the retrieval of approximate reflectance spectra regardless of the sensor having imprecise radiometric or wavelength calibration and unknown solar illumination intensity (Bernstein *et al.,* 2005). This calibration method is essentially suitable for the normalization of multiple scenes typical of many temporal monitoring studies, due to its high computational speed in comparison to first principles algorithms such as the Empirical Line Method.

The QUAC algorithm uses the scene parameters to retrieve surface reflectance and the mathematical formulation is shown below.

$$\rho_j^o(\lambda) = \frac{\rho_j(\lambda) - \rho_b(\lambda)}{g_o \sigma\rho(\lambda)} \qquad (1).$$

Where λ is the wavelength

ρ_j^o is the spectral standard deviation for a group of diverse materials which is a nearly wavelength independent constant.

g_o is the normalization factor and $\sigma\rho$ is the correction factor.

ρ_b is the base line contribution.

The application of satellite imagery of different spatial resolution for change detection is problematic. In post-classification change detection however, the thematic resolution of the classified maps affects the change detection results (Buyantuyev and Wu, 2007). In this study the mismatch between the spatial resolution of Landsat MSS and Landsat 5 TM was solved by resampling the Landsat MSS imagery to 30m. Resampling to a higher resolution firstly ensures a pixel overlay between Landsat MSS and Landsat 5 TM and secondly maintains the radiometric fidelity of Landsat MSS. Wickham and Rhtters (1995) indicate that landscape metrics are not significantly affected by the change in pixel size up to 80m if the land cover classifications are generated by sensors with different spatial resolving powers such as Landsat TM and Landsat MSS. To further resolve the uncertainties that arise due to different thematic resolutions of the classified maps, an object-oriented classification based on multiresolution segmentation was implemented. Multiresolution segmentation merges similar pixels into homogeneous objects that are suitable for landscape pattern analysis. The thematic resolution of the classified maps, are less affected when objected-oriented classification is used to produce maps of high accuracy (Shao and Wu, 2008).

3.2.3 Classification Method

Image classification was executed through an object oriented classification algorithm called the Standard Nearest Neighbour Classification. This is essentially a supervised classification technique which uses selected image objects as training data. The Landsat images were segmentated into image objects using a multiresolution image segmentation algorithm. The algorithm incorporates both spectral and spatial information in the image segmentation phase resulting in meaningful image objects which carry typical characteristics of the land covers as compared to pixels. The homogeneity criterion of the multiresolution segmentation algorithm measures how homogeneous or heterogeneous an image object is within itself.

Landsat TM images were partitioned into image objects using Definiens Developer software package (Definiens, 2007). Generation of image objects was achieved through an image multiresolution segmentation procedure in Definiens. Decisions regarding selection and weighting of inputs to the segmentation process were based on the spectral and spatial characteristics of the individual Landsat bands and experimentation. The three visible bands were assigned equal weightings, and the sum of the weightings assigned to all the bands combined equalled that assigned to the near-infrared band. Each input scenario was evaluated on its ability to delineate meaningful landscape components based on the visual inspection of the segmentation output (Chubey *et al.*, 2006).

The Nearest Neighbour classification method was used to classify image objects based on user-defined functions of object features. The method uses a set of samples for different classes in order to assign membership values. Anderson *et al*. (1976) highlight that there is no one ideal classification of land use and land cover, the process is subjective depending on the different perspectives in the classification process. In this study a total of five classes were chosen for the classification *viz*: intact vegetation (V), degraded vegetation (DV), settlements (S), bare and degraded soil (BDS) and water (W). The intact vegetation category included natural forests, grass, crops, shrubs, riparian vegetation, and plantations. Degraded vegetation refers to vegetation which has lost the structure, function, species composition and productivity generally associated with intact native vegetation and is predominate in disturbed areas (ITTO, 2002). Examples of degraded vegetation constituted *Acacia Karroo*, *Aloes*, *Pteronia incana* and very sparse vegetation. Figure 3.1 shows photographs acquired in the central part of the catchment, which illustrate vegetation species and conditions which were classified as degraded vegetation. Bare and degraded soil includes barren areas and damaged soil that has been affected by erosion or depleted of nutrients. The water class included water bodies in the catchment like rivers, dams and the estuary. Sample training points for the different land use/cover types were collected in the field using a GPS to facilitate supervised classification.

Figure 3.1 Degraded vegetation.

Accuracy assessment was done for all the classification results by first identifying features on the satellite imagery from 1972 to 2006 that could still be identified in the field. This involved collecting stable land cover features such as nature reserves, forestry plantation, mature mixed forest, dams, centroids of settlements that were in existence since 1972. In addition to image analysis, degraded rangelands were also established using aerial photographs. Acentimetre level precision Ashtech®ProMark2™ Global Positioning System (GPS) receiver was then used to validate these reference points in the field. At least 70 ground reference points per class were collected, the number of points was controlled by class size. Additional points were also collected from a 2,5m spatial resolution pansharped SPOT image acquired in December 2006. The GPS coordinates were converted to shapefiles before exporting them to Definiens Developer 7 software for sample extraction. Ground based sample points were superimposed on a segmented satellite imagery to facilitate extraction of pixels for use as test areas.

3.2.4 Landscape Fragmentation

Landscape fragmentation analysis was performed in communal areas in central Keiskamma catchment as well as the riparian and hillslope zones of the catchment. Delineating the spatial extent of the riparian and slope river proximal zone is problematic and is subject to a lot of ecological and geomorphologic debate (Muller, 1997; Naiman and De'camps, 1997). The riparian zones were extracted from the object oriented classified images using a fixed width one kilometre buffer generated around the rivers in ArcView 3.3. Central Keiskamma communal areas were extracted from the catchment using PCI Geomatica 10.1 software to permit more detailed landscape structural analysis. Patch Analyst, a FRAGSTATS interface in ArcGIS was used to compute class level landscape metrics to analyse landscape structure and its change over time within riparian hillslope zones and communal areas. Class level metrics are useful fragmentation indices because they measure the quantity and distribution of a particular land use/cover class (McGarigal *et al.*, 2002). In this study eight class level landscape metrics that were considered effective in determining landscape structural changes were selected for fragmentation analysis. These are a) number of patches (NUMP) (b) edge density (ED) (c) class area (CA) (d) mean shape index (MSI) (e) mean nearest-neighbour metric (MNN) (f) mean proximity indices (MPI) (g) mean patch size (MPS) and (h) interspersion and juxtaposition index (IJI). Selection of the metrics was based on the scale of the analysis and relevance for monitoring riparian zone and hillslope degradation. A detailed description and interpretation of the landscape metrics is provided by McGarigal *et al.* (2002).

Characterisation of landscape organisation using the Landscape Functional Analysis (LFA) technique (Tongway and Hindley, 2004a) is an effective means to validate rangeland condition and landscape fragmentation indices, which relate to the connectivity of vegetation patches. The landscape organisation index (LOI) is defined as the proportion of length of patch to the total length of transect (Tongway and Hindley, 2004b). A totally bare transect will have an index of 0 while transects filled with patches will have an index of 1 (Tongway and Hindley, 2004b). Fieldwork was conducted to collect landscape organisation data using line transects oriented along the maximum slope direction of hillslopes. Twelve transects per rangeland type were generated with the aid of classified imagery in the former commercial farms, highly degraded communal areas and communal areas with good rangeland condition in order to validate the differences in rangeland condition. A continuous record of patch/inter-patch distances were collected along transects as a means of characterising landscape organisation. Analysis of variance (ANOVA) single factor statistics ($\alpha = 0.01$) was used to test for significant differences between the mean landscape organisation indices obtained in former commercial farms, degraded communal areas and communal areas in pristine condition. The current condition of riparian zone and adjacent hillslopes was validated by means of the Rapid Assessment of Riparian Condition (RARC) technique at 10 sites along a reach in the communal areas section of the Keiskamma River. A more detailed assessment of riparian vegetation along the Keiskamma River was done by Rowntree (1991), Colloty (1997), Hall (1997), and Matoti (1999). Their work provides more detailed field evidence of riparian and adjacent hillslope vegetation condition.

3.3 Results

Land cover classification maps for 1972 and 2006 are presented in Figures 3.3 up to 3.8. Validation of the classification results proved that the object-oriented classification produced valid and reliable land cover maps since all the overall accuracies were higher than 0.819 and the Kappa Index of Agreement (KIA) was above 0.749. A full error matrix is presented in Appendex A. The separation of settlements and bare and degraded soils was not easy due to their spectral similarity in communal areas and the low spatial resolution of Landsat MSS imagery. This resulted in lower classification accuracies for these classes (Appendix A). A summary of the overall accuracies and KIA are presented on Table 3.1.

Table 3.1 Accuracy assessment summary (1977-2006)

Year	***Overall Accuracy***	**KIA**
1972	0.861	0.762
1988	0.819	0.749
1992	0.839	0.780
1997	0.899	0.867
2001	0.920	0.892
2006	0.898	0.866

3.3.1 Land use/cover changes between 1972 and 2006

A change detection analysis was conducted to determine the land use and cover trends that have occurred in the Keiskamma catchment. The changes mapped are quantified in the clustered column graph (Figure 3.2). The trends generally show increasing degradation from 1972 to 2006. This change is however non-linear; the trends indicate cyclic transitions of decline and recovery in vegetation cover. The 1993 classification shows large bare patches within the forest plantation along the Amatole mountain range, which could be a result of harvesting in the pine plantations. In contrast to intact vegetation, degraded vegetation shows a general increase between 1972 and 2006. The 1988 classification also shows a proliferation of bare and degraded soil patches which tend to merge, forming bigger bare soil surfaces in subsequent years. The overall land use/cover transformation that occurred in the Keiskamma catchment between 1972 and 2006 are shown by the change detection matrix Table 3.2 . The intact vegetation class changed to Degraded Vegetation by 528.792km^2 and to Bare and Degraded Soil by 39.746km^2, marking a net vegetation decrease during this period. The Degraded Vegetation class in particular increased by 194.952km^2. Further degradation is also noted on the Degraded Vegetation class to Bare and Degraded Soil by 139.822km^2. The Bare and Degraded Soil Class increased by 179.322km^2. The trends however show that bare and degraded soil has the potential to recover, as revealed by the conversion of

12.182km^2 to bare and degraded vegetation and a further 6.383km^2 to vegetation. The overall trends however reveal increasing degradation, as manifested by increases in degraded vegetation, bare and degraded soil.

Table 3.2 Change detection statistics 1972 to 2006.

Final State 2006	InitialState 1972- Area (km^2)					
	V	W	DV	S	BDS	Total
V	626.1435	2.6703	232.1289	19.2735	6.3828	886.599
W	3.0519	0.3843	2.7387	0	0.0009	6.1758
DV	528.7923	3.7656	907.8786	66.2949	12.1824	1518.914
S	18.9333	0.0783	41.3937	8.6085	3.0087	72.0225
BDS	39.7458	0.7038	139.8222	20.6253	8.2026	209.0997
Total	1216.6668	7.6023	1323.9621	114.8022	29.7774	2692.811
Class Difference	-330.0678	-1.4265	+194.9519	+42.7797	+179.3223	0

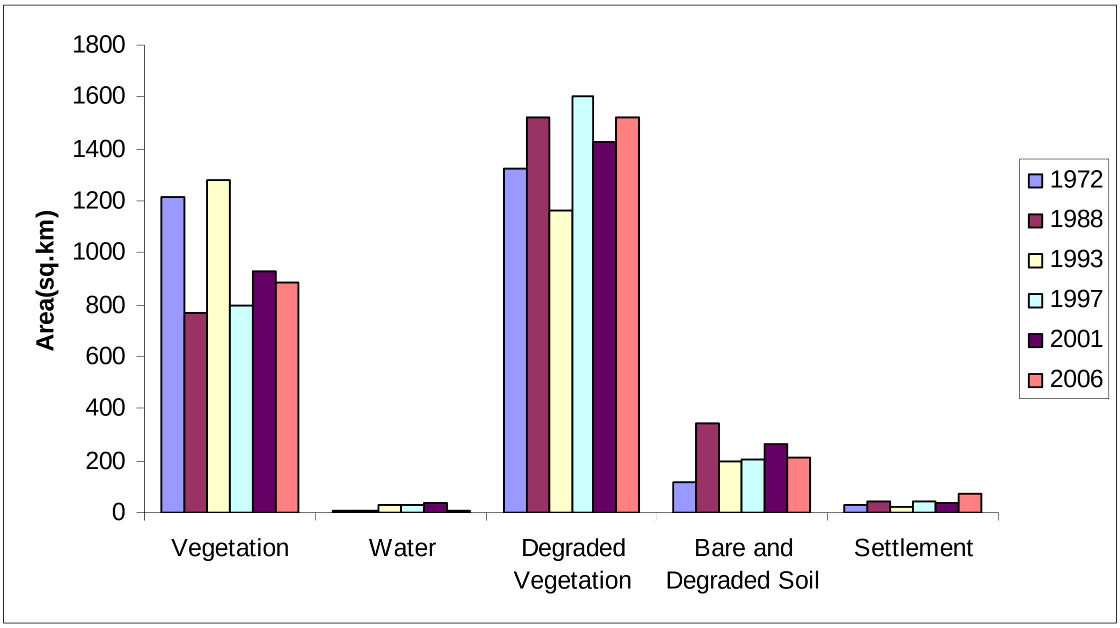


Figure 3.2 Land use and cover trends 1972 to 2006.

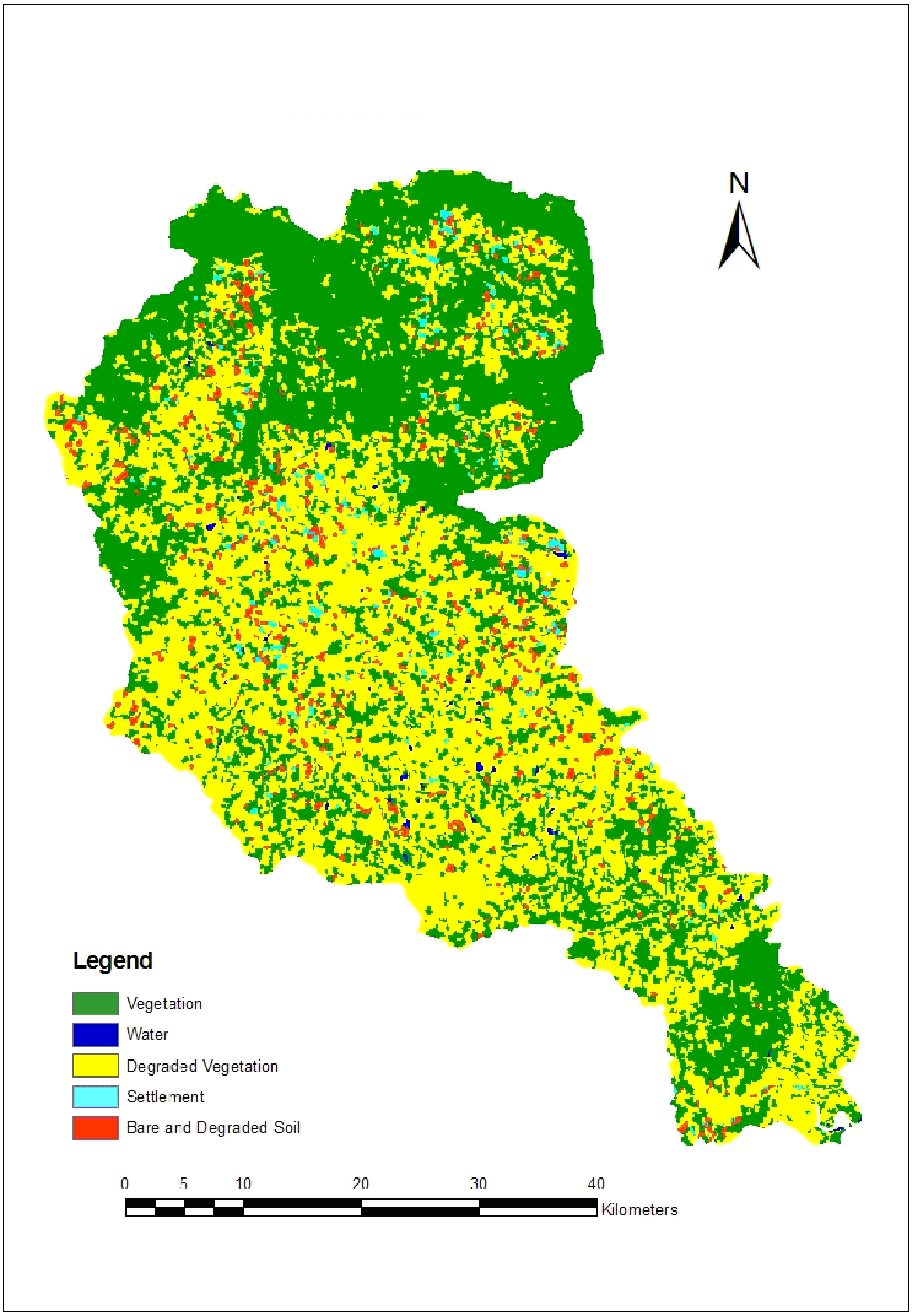


Figure 3.3 1972 LULC Classification.

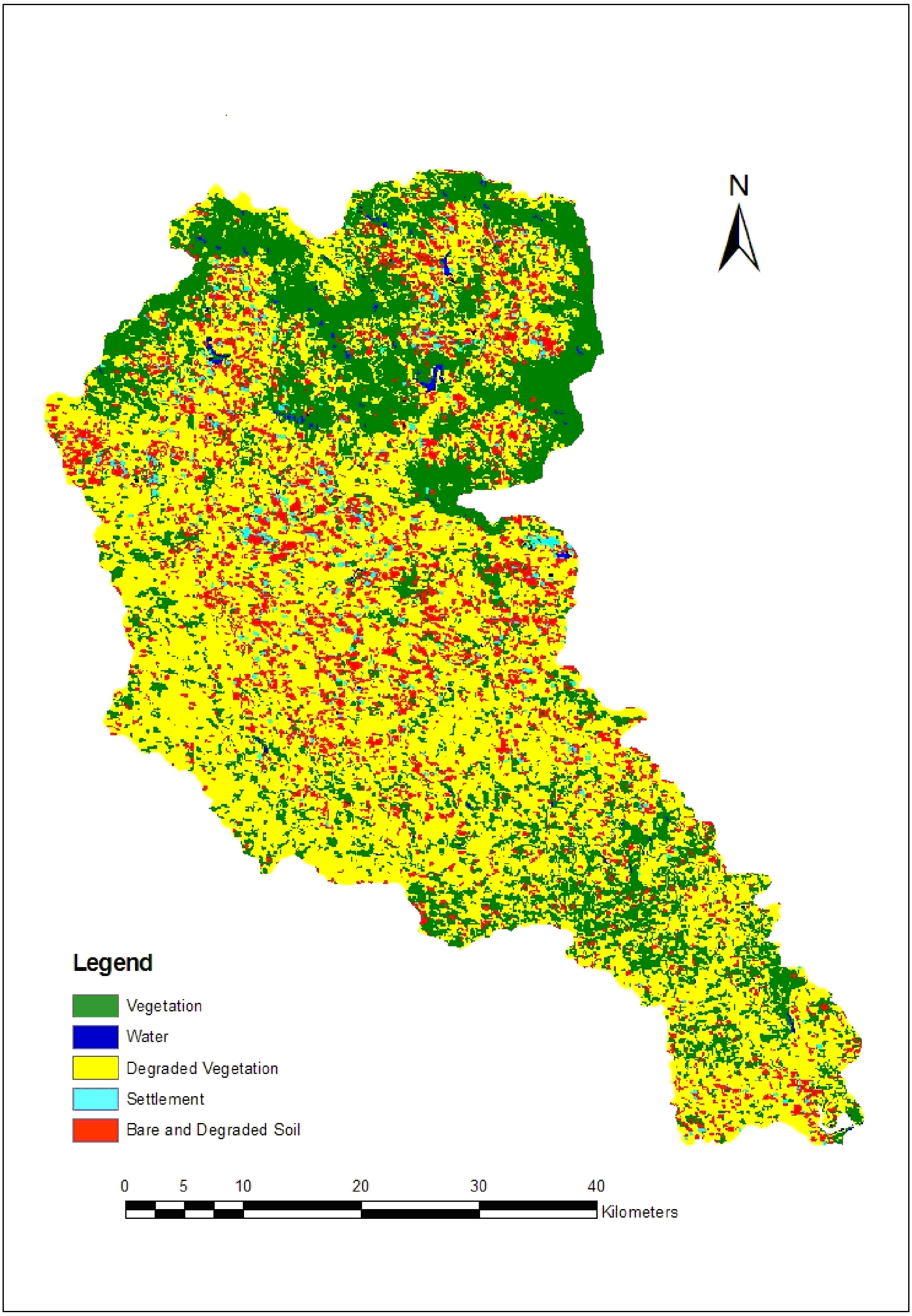


Figure 3.4 1988 LULC Classification.

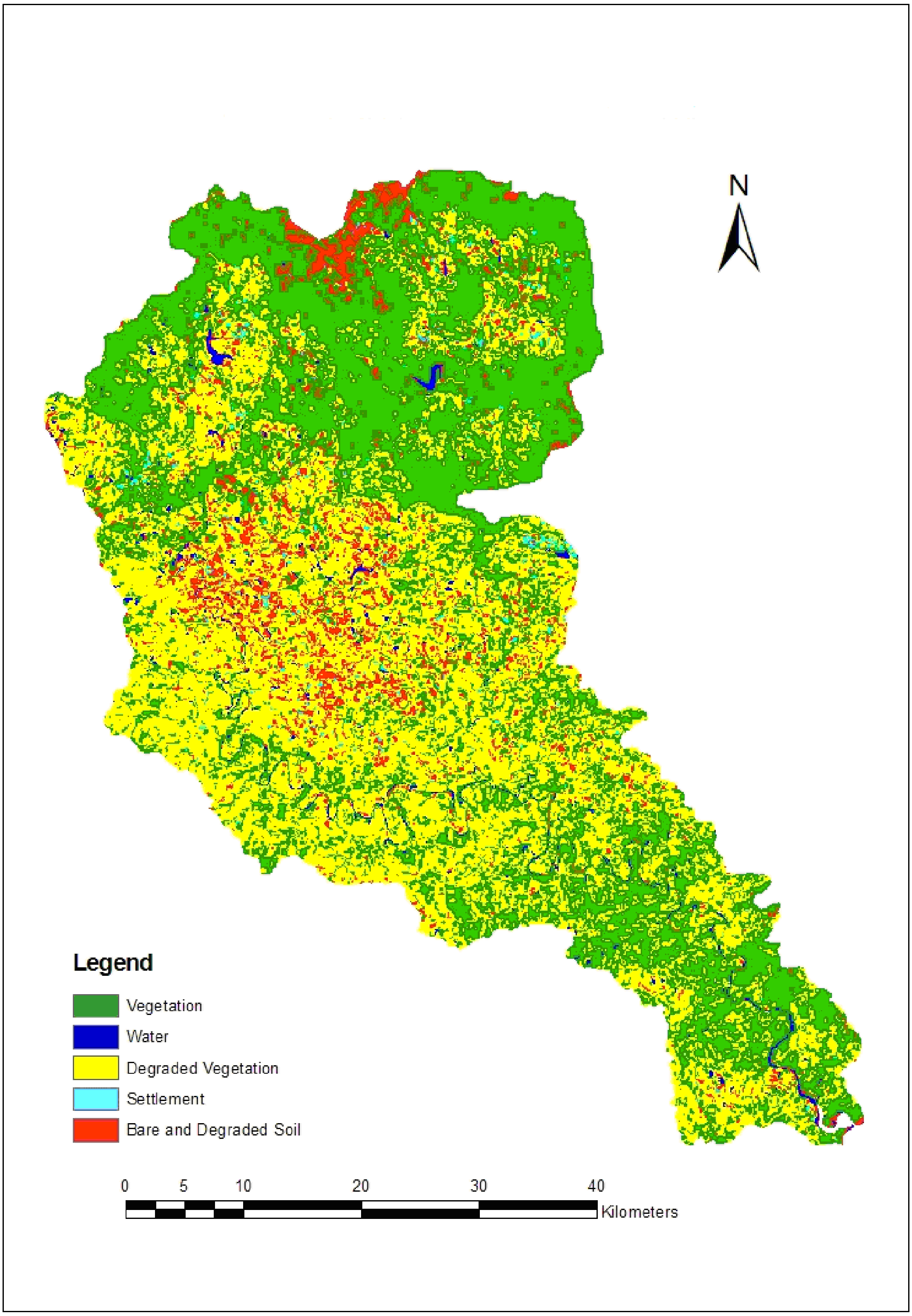


Figure 3.5 1993 LULC Classification

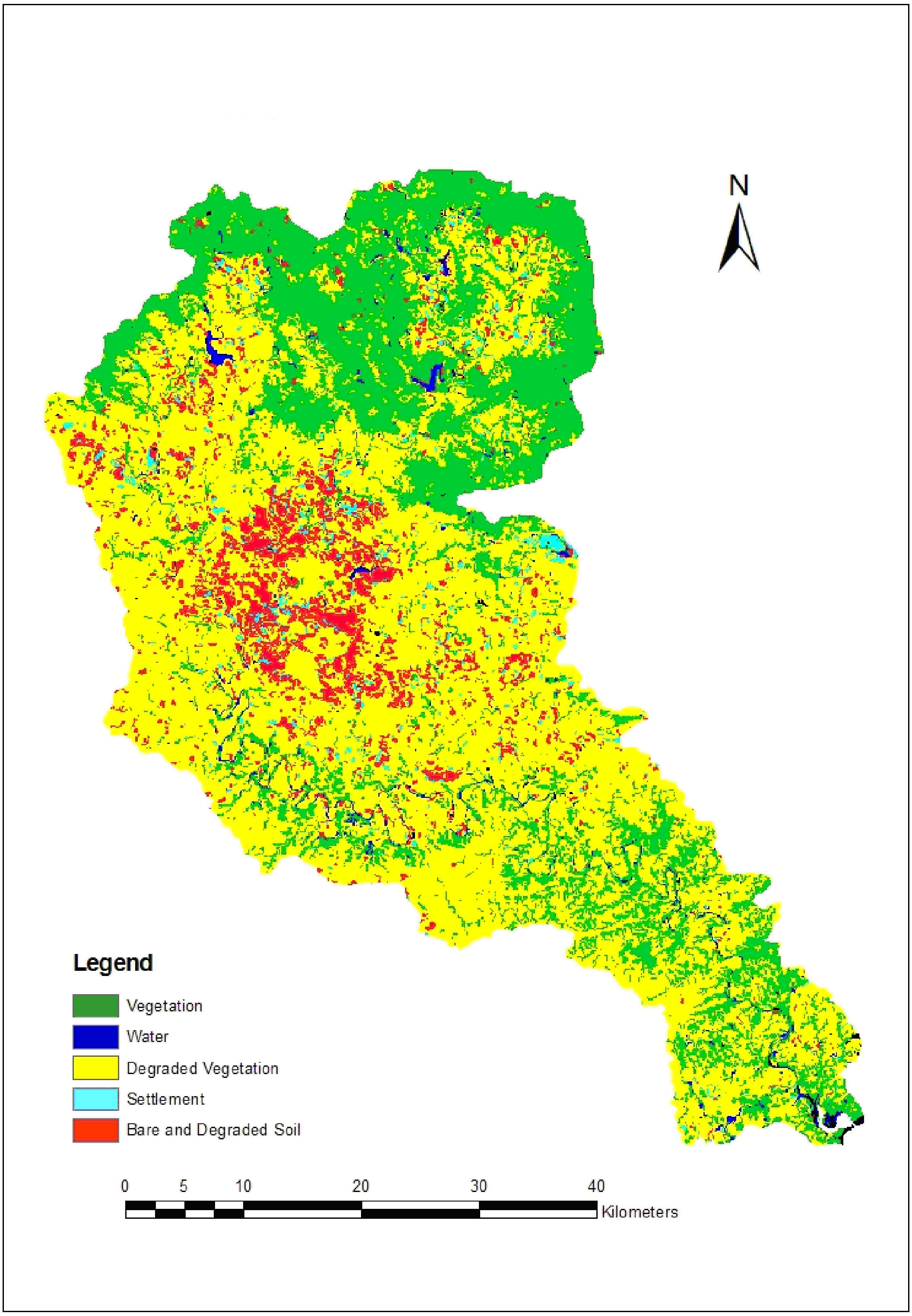


Figure 3.6 1997 LULC Classification.

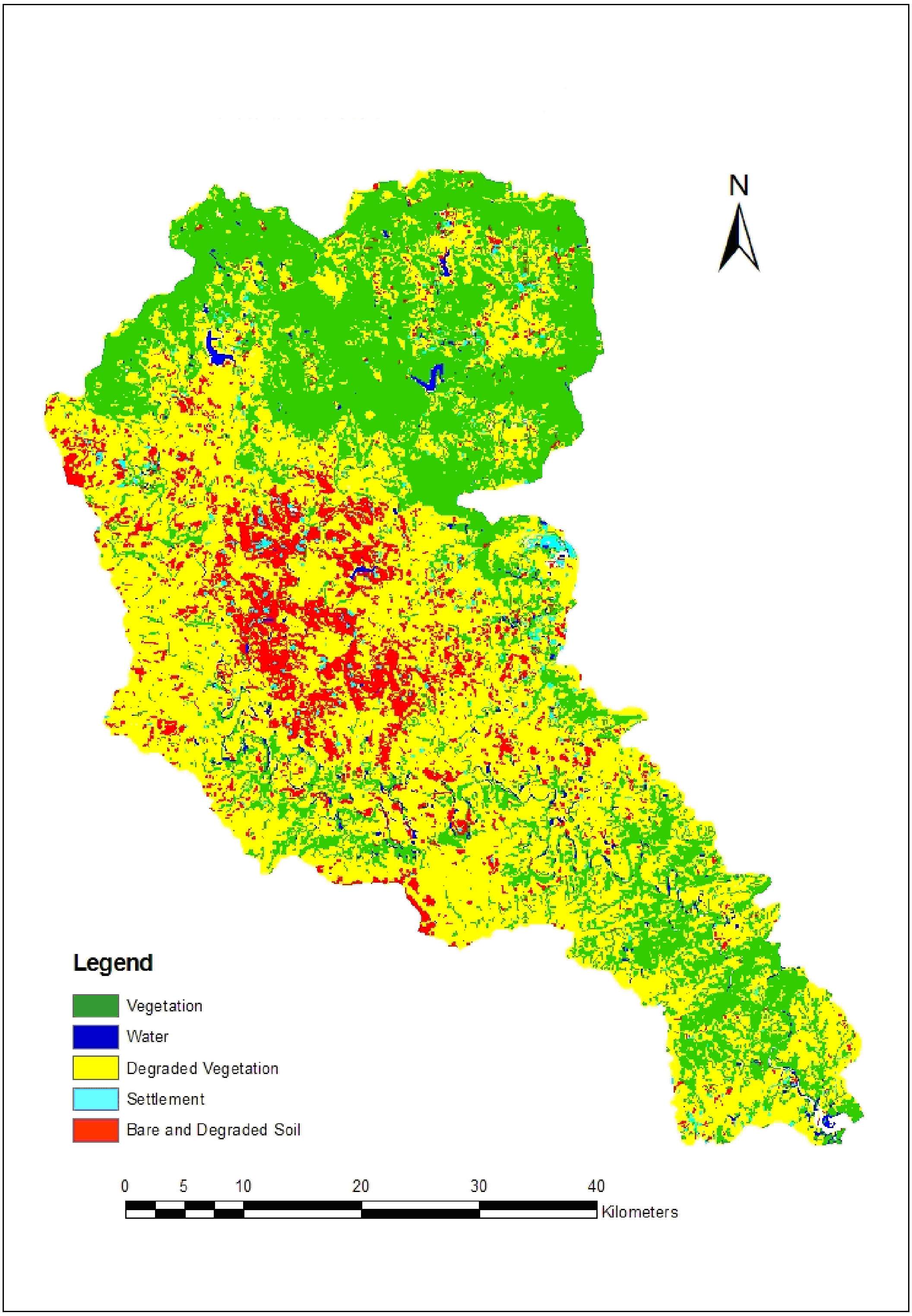


Figure 3.7 2001 LULC Classification.

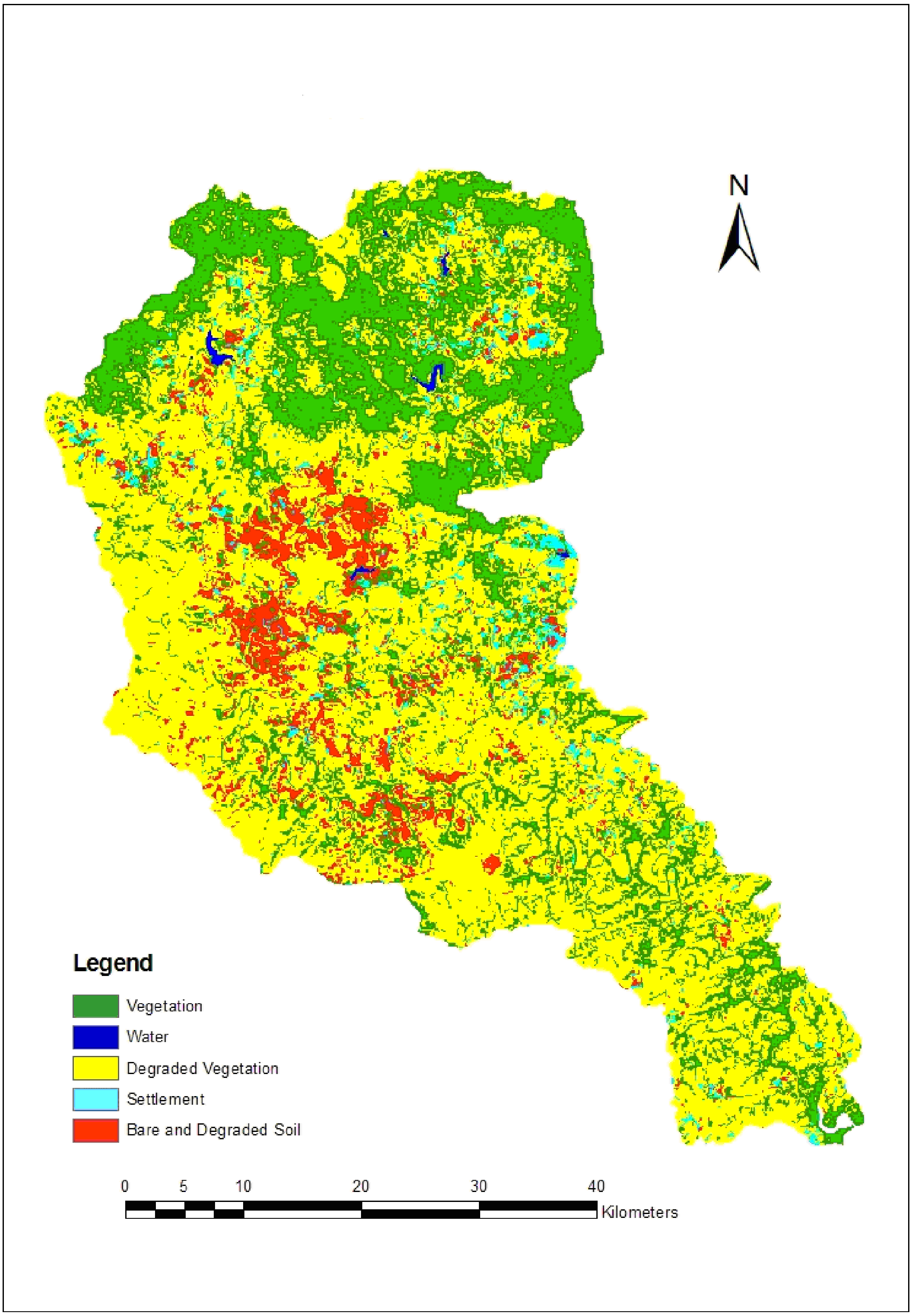


Figure 3.8 2006 LULC Classification.

3.3.2 Fragmentation Analyses of the Riparian and Proximal Hillslopes

The results of the analysis of the riparian and proximal hillslope are presented in Table 3.3. The land cover status for the riparian and proximal hillslopes in 1972 and 2006 are shown in Figure 3.9.

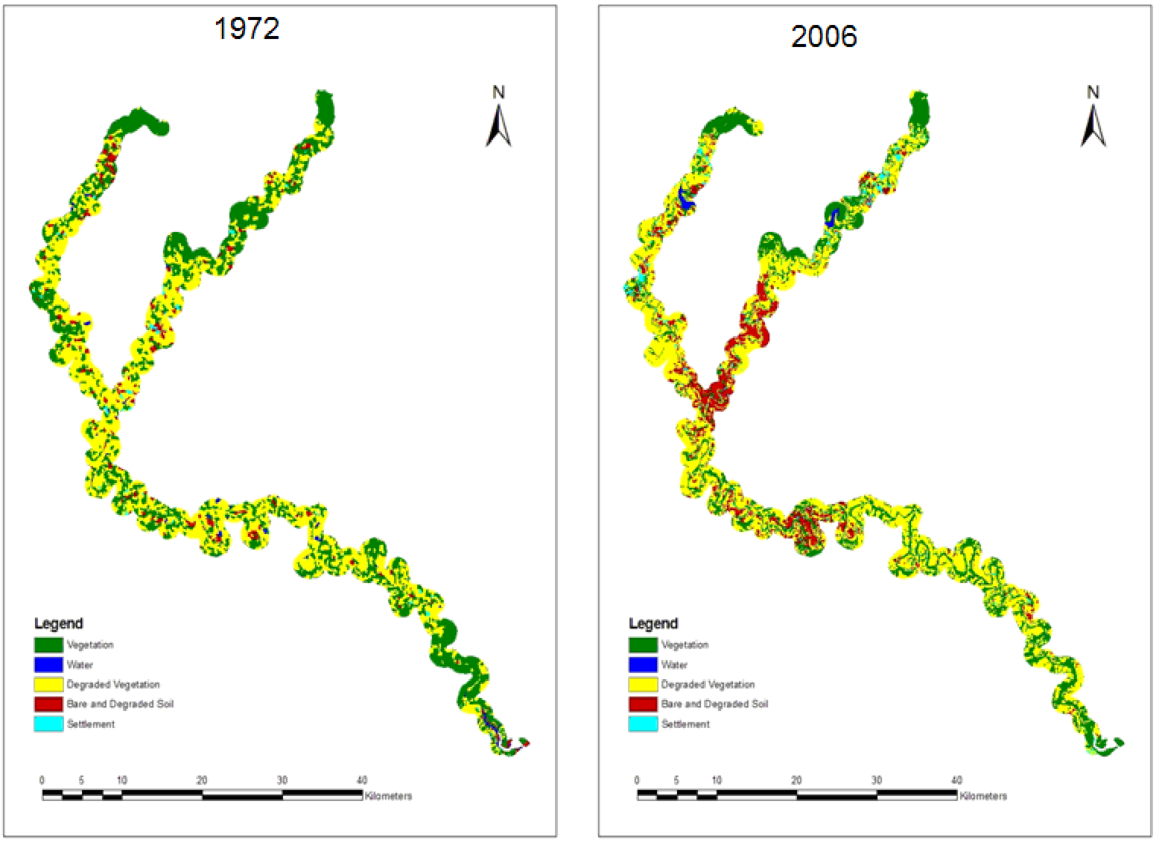


Figure 3.9 1972 and 2006 riparian and hillslope zone condition

The class landscape metric indicates that the riparian and hillslope vegetation was more fragmented in 2006 compared to 1972. This is revealed by the increase in the number of patches and edge density of vegetation in the riparian and adjacent hillslope zone. The number of patches increased from 322 to 531 while the edge density increased from 3.38 m/ha to 4.33 m/ha. These landscape metrics indicate that the riparian and hillslope vegetation was fragmented into smaller patches. The mean patch size decreased from 73.95ha in 1972 to 31.86ha in 2006, signifying a decrease in vegetation patch size. The mean shape index increased from 1.67ha to 1.77ha between 1972 and 2006 indicating that the fragmentation is not directly linked to human activities such as systematic demarcation of land for cultivation and development of plots. It could therefore be inferred that degradation in the riparian and hillslope zones could be a result of overgrazing and loss of saturation.

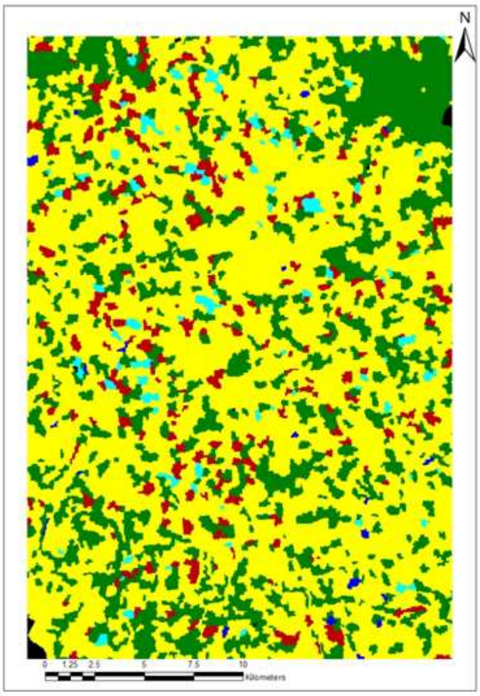
Table 3.3 Landscape metrics change at the patch class level and riparian-hillslope zone.

Class	*NUMP*	*ED*	*CA*	*MPS*	*MSI*	*IJI*	*MPI*	*MNN*
1972								
Vegetation	322.00	3.38	23812.02	73.95	1.67	49.44	886.31	162.43
Water	40.00	0.16	471.41	11.81	1.53	75.54	1.64	1975.34
Degraded Vegetation	201.00	3.72	28741.05	142.99	1.78	58.97	6473.30	192.72
Bare and Degraded Soil	186.00	0.71	2758.68	14.83	1.41	68.36	18.60	507.19
Settlement	51.00	0.16	512.46	10.05	1.34	71.74	8.58	1357.10
2006								
Vegetation	513.00	4.33	16919.10	31.86	1.77	44.72	3087.09	126.61
Water	11.00	0.06	388.35	35.30	1.60	26.91	0.94	9159.48
Degraded Vegetation	377.00	5.79	30336.12	80.47	2.01	59.52	4831.27	83.70
Bare and Degraded Soil	403.00	2.05	7339.77	18.21	1.64	49.98	361.51	198.05
Settlement	176.00	0.60	1326.87	7.54	1.66	66.09	24.97	500.56

Unlike the intact vegetation class, the degraded vegetation class became more interconnected and clumped. This is portrayed by the increase in the Interspersion and Juxtaposition indices which changed from 58.97% in 1972 to 59.52% in 2006. Although the number of patches and edge density increased for degraded vegetation, the class area increased from 28 741.05ha to 30 336.12ha indicating an increase in degraded vegetation within the riparian and adjacent hillslope zones. The mean nearest neighbour distance for degraded vegetation patches decreased from 192.72m to 83.70m, signifying that degraded vegetation patches were merging. Bare and degraded soil class also shows significant increases in the class area from 2 758.68ha in 1972 to 7 339.77ha in 2006. The mean proximity index and mean nearest neighbour distance also show that bare and degraded soil patches become less isolated and more interconnected to each other. These results indicate increasing land degradation in the riparian and hillslope zones. Processes such as soil erosion are evident on river banks and hillslope proximal zones of the Keiskamma river.

3.3.3 Fragmentation Analyses of the central Keiskamma catchment

The results of the fragmentation analysis of the central Keiskamma catchment are presented in Table 3.4 while the classified images for the communal villages of the central Keiskamma catchment for 1972 and 2006 are shown in Figure 3.10 and Figure 3.11 respectively. Fragmentation assessment of the central Keiskamma catchment indicates that vegetation cover has become more fragmented in 2006 than in 1972. This is indicated by increases in the number of vegetation patches, which rose to 610 in 2006 from 372 in 1972. This shows that large vegetation patches were broken up into many but much smaller vegetation patches. The mean patch size for vegetation decreased from 47.62ha in 1972 to 20.50ha in 2006, indicating a reduction in patch size due to fragmentation. Increasing fragmentation in the central Keiskamma is highlighted by the increase in vegetation patch edge density, which increased from 22.03 m/ha in 1972 to 27.41 m/ha in 2006. Further evidence of fragmentation in the central Keiskamma is revealed by the reduction in the class area for vegetation which decreased to 12506.67ha in 2006 from 17716.23ha in 1972.



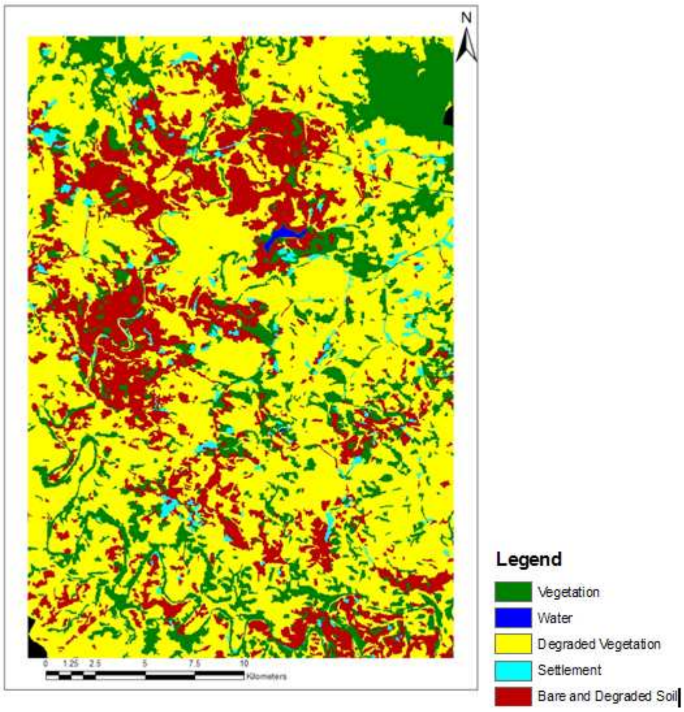


Figure 3.10 1972 Central Keiskamma area **Figure 3.11 2006 Central Keiskamma area**

The mean shape index for vegetation increased from 1.54 in 1972 to 1.76 in 2006. This indicates that vegetation patches became more geometrically complex in 2006 compared to the situation in 1972. A mean shape index value closer to 1 indicates that the patches are more regularly shaped. The IJI which measures patch adjacency increased from 25.56% in 1972 to 41.40% in 2006, implying more interspersion between vegetation patches. The mean proximity index for vegetation patches also increased for the period under review.

Table 3.4 Landscape metrics change at patch class level, central Keiskamma.

Class	*NUMP*	*ED*	*CA*	*MPS*	*MSI*	*IJI*	*MPI*	*MNN*
1972								
Vegetation	372.00	22.03	17716.23	47.62	1.54	25.56	244.42	205.84
Water	22.00	0.53	215.01	9.77	1.35	43.42	0.08	2806.85
Degraded Vegetation	27.00	27.83	44200.53	1637.06	2.26	50.03	34440.03	273.59
Bare and Degraded Soil	217.00	7.33	4011.84	18.49	1.38	43.46	8.63	470.07
Settlement	79.00	2.39	1263.06	15.99	1.33	54.71	1.87	989.76
2006								
Vegetation	610.00	27.41	12506.67	20.50	1.76	41.40	456.72	162.21
Water	3.00	0.16	80.19	26.73	2.05	14.58	0.00	9050.81
Degraded Vegetation	242.00	44.12	39451.32	163.02	1.86	55.41	105877.20	74.17
Bare and Degraded Soil	511.00	25.86	1447.56	27.28	1.69	43.56	1077.38	160.74
Settlement	189.00	5.26	13941.54	7.66	1.78	63.97	16.57	416.45

Increased fragmentation of degraded vegetation is revealed by increases in the number of patches, edge density and decreases in mean patch size. The fragmentation assessment also proves that the mean patch size for bare and degraded areas increased to 27.28ha in 2006 from 18.49ha in 1972, implying a considerable increase in degradation. The largest patch index for bare and degraded soil increased to 3.30ha in 2006 from 0.14ha in 1972, which gives an indication of the expansion of bare and degraded soils. Increases in the edge density for bare and degraded soils also attest to increasing degradation trends. A reduction of 470.07m to 160.74m in the mean nearest neighbour distance indicates that degraded bare soil patches are less isolated from each other.

Image analysis and field validation indicate contrasting rangeland conditions in the different communal villages and former commercial farms. These contrasts are particularly evident between villages managed by different traditional institutions that are separated by road and fence boundaries. Figure 3.12 shows highly degraded communal villages close to Peddie town while villages around Koloni shown on Figure 3.13 have more intact vegetation condition. Field observations in the communal areas confirmed the vast tracts of sparse and degraded vegetation. Many hillslopes bordering the riparian zone in communal settlements are characterised by gully erosion and invader vegetation types indicative of degradation such as *Acacia Karroo* and *Pteronia incana* patchy shrub of karroid origin.

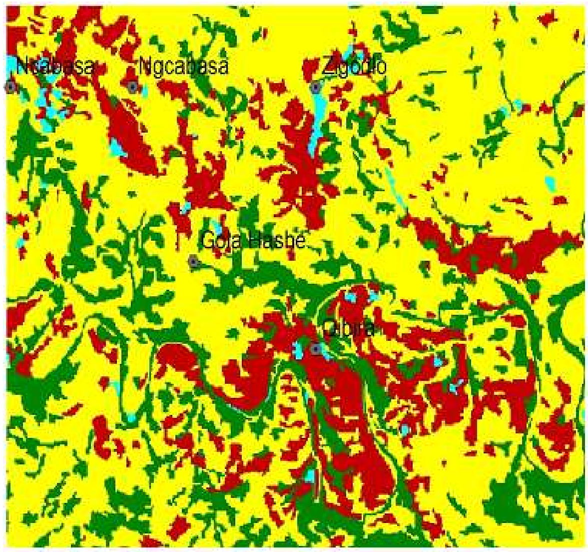


Figure 3.12 Degraded communal villages (2006)

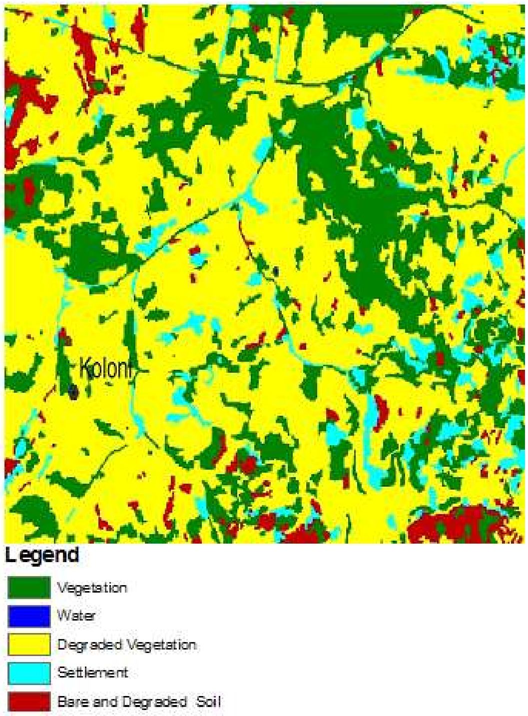


Figure 3.13 Pristine communal villages (2006)

Abandonment of agricultural fields, evidence of which can be seen in the field in the form of eroded contours, gullies and blanket invasion by *P. incana* and *Acacia Karroo,* as well as crusted soil surfaces is a widespread phenomenon.

3.3.4 Landscape Function Analysis

Landscape organisation indices in degraded communal rangelands, pristine communal rangelands and former commercial farms are different as indicated by the *P-value* of 1.5×10^{-5} that is less than the significance level of 0.01. The variances in all three different sites are not equal as confirmed by *F* (15.850) which is greater than *F crit* (5.132). A summary of the ANOVA analysis is shown in Table 3.5 and Table 3.6. A comparison between the degraded communal rangelands and pristine communal rangelands indicates that there is a significant difference between them. The average landscape organisation index for degraded communal rangelands is 0.268 while pristine communal rangelands have a mean landscape organisation index of 0.509. The summary of ANOVA statistics provided in Table 3.8 reveal differences between degraded and pristine communal rangelands, this is shown by a *P-value* (5.560×10^{-6}) which is less than the significance level of 0.01 and *F* (35.327) which is greater than *F crit* (7.945). A comparison between pristine communal rangelands and former commercial farms indicate that while former commercial farms have a slightly higher landscape organisation index of 0.536 compared to 0.509 in pristine communal rangelands. This difference is not significant as shown by the ANOVA analysis results on Table 3.7. The similarity in landscape organisation between pristine communal areas and former commercial farms is reflected by a *P-value* (0.652) that is greater than the significance level 0.01 and *F* (0.209) that is less than *F crit* (7.945) (See Table 3.8). The differences in the characterisation of landscape organisation are illustrated on a box and whisker diagram on Figure 3.14.

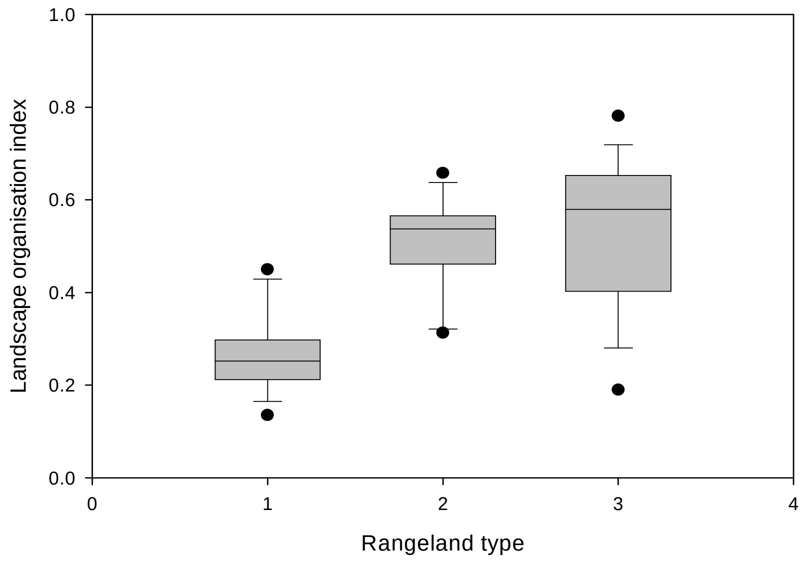


Figure 3.14 Box and whisker diagram showing landscape organisation index variability.

(1 = degraded communal rangeland; 2 = pristine communal rangeland; 3 = former commercial farms).

Table 3.5Anova Summary

Group	*Count*	*Sum*	*Average*	Variance
Degraded Communal	12	3.221	0.268	0.008
Pristine Communal	12	6.11	0.509	0.011
Former Commercial Farms	12	6.432	0.536	0.030

Table 3.6 ANOVA: Degraded Communal, Pristine Communal and Former Commercial Farms.

Source of variation	*SS*	*df*	*MS*	*F*	*P-value*	***F crit***
Between Groups	0.521	2	0.261	15.85	1.5E-05	5.321
Within Groups	0.543	33	0.016			
Total	1.064	35				

Table 3.7 ANOVA: Degraded Communal and Pristine Communal.

Source of variation	*SS*	*df*	*MS*	*F*	*P-value*	***F crit***
Between Groups	0.348	1	0.348	35.327	5.56E-06	7.945
Within Groups	0.216	22	0.010			
Total	0.564	23				

Table 3.8 ANOVA: Pristine Communal and Former Commercial Farms.

Source of variation	*SS*	*df*	*MS*	*F*	*P-value*	***F crit***
Between Groups	0.004	1	0.004	0.209	0.652	7.945
Within Groups	0.451	22	0.020			
Total	0.455	23				

The results of the current condition of riparian and hillslope proximal zones are shown in Table 3.9. The rapid assessments indicate that fragmentation in the riparian and the hillslope proximal zones are mainly as result of overgrazing, cultivation, deforestation and settlement. Vegetation cover within riparian zones was relatively dense and almost pristine, a remarkable contrast however, occurs immediately after this boundary, the hillslope a severely degraded and predominately comprised of xeric plants. This scenario is vividly shown in the photographs (Figure 3.15) acquired in October 2010. Patches of Acacia karroo occurring outside the riparian zone were observed above the Sandile dam. *Cynodondactylon* and *Acacia karroo* were dominant outside the riparian vegetation.

Figure 3.15 Degraded hillslope vegetation adjacent to intact riparian vegetation.

Table 3.9 Rapid assessment of riparian condition

Site No	Channel width(m)	Riparian width (m)	Riparian vegetation cover (%)	Hillslope vegetation cover (%)	Evidence of fragmentation
1	16	24	90	30	Cultivation, settlement, cattle tracks, foot paths
2	10	15	90	20	Cultivation, settlement, cattle tracks, deforestation
3	7	3 6	85 85	10 -	Cultivation, settlement, foot paths, deforestation.
4	10	12	90	15	Cultivation, settlement, foot paths, deforestation, rill and gully erosion.
5	10	25	95	15	Cultivation, settlement, footpaths, overgrazing.
6	20	21	100	60	Patch vegetation
7	9	12	57	43	Degraded riparian forest
8	10	15	72	59	Intact
9	10	12	89	78	Grass dominated riparian forest.
10	15	20	60	20	Hillslopes comprised of degraded vegetation species. (Acacia karroo)

3.4 Discussion

The object oriented approach produced a smooth classification devoid of the mixed pixel effect. The conceptualization of image objects as ecological units was useful for fragmentation analysis since their patterns and structure could be monitored through time. While the ultimate analysis indicates that intact vegetation has undergone a significant decline from 1972 and 2006, the temporal changes observed in the intermediate years suggest that vegetation in the Keiskamma catchment undergoes cyclic transitions of decline and recovery. The overall results indicate a decline in intact vegetation cover, an increase in degraded vegetation and bare eroded soil. Severe gully erosion on abandoned lands and vegetation invasion by dwarf shrubs are ubiquitous phenomena in many communal areas of the Eastern Cape Province (Kakembo and Rowntree, 2003). Severe rill and gully erosion were observed on overgrazed and abandoned lands during field visits.

Long term land cover change detection is inherently difficult due to constraints imposed by vegetation phenology, seasonality and variability in inter-annual vegetation productivity (Coppin *et al*., 2004). In this study the effect of rainfall variability on net land use/cover changes has not been considered. Whilst the importance of that is known, the post-classification comparison technique used in this study compensates for inter-date phenological variations since each classification is generated independently and the different classifications are then used to characterize the land-cover transformations (Coppin *et al*., 2004; Rogan *et al.* 2003; Yuan *et al*., 2005). In addition the use of post-classification comparison techniques was beneficial in detecting change using multi-temporal satellite imagery of different spatial and spectral resolutions. Sing (1989) and Rogan *et al*. (2003) provide a more detailed overview of the advantages post-classification change detection.

The disadvantage of object-oriented post-classification change detection method implemented in this study is that it does not permit detection of very small changes within land use/cover classes. Although this problem is also present in per-pixel post-classification change detection (Rogan *et al*., 2003) , it is amplified in object-based methods since similar pixels are merged together to form objects thus reducing the chances of detecting subtle changes within land cover classes.

The fragmentation analysis in the riparian zone indicated a reduction in intact vegetation. Rapid assessment of the current condition of the riparian and proximal hillslope zones indicate that fragmentation in these areas is directly caused by anthropogenic activities such as overgrazing, deforestation, cultivation and settlement. Whereas the riparian zones remained relatively intact, the proximal hillslopes are highly degraded with very low vegetation cover. It could also be inferred that the reduction in intact native vegetation is a result of loss of saturation in the riparian and the proximal hillslopes induced by impoundments. The presence of xeric plants and other invasive plant species within the adjacent hillslopes could be linked to post impoundment hydrological changes; further evidence is required to ascertain this possibility. This condition is undoubtedly exacerbated by anthropogenic induced degradation of the riparian and hillslopes zones. A reduction in the plant water requirements could stress native vegetation and lead to the successive dieback of older plants while inhibiting regeneration of younger pioneer species (Nilsson and Berggren, 2000). Increases in degraded vegetation within the riparian and proximal hillslope could be ascribed to the loss of saturation in the hillslopes. Nilsson and Berggren (2000) observe that riparian zones are generally vulnerable to invasion by exotic species because rivers are dynamic and have recurrent disturbances, which are more pronounced in regulated rivers. The reduction in flow and changes in the pattern of flooding also provide ideal conditions for the establishment of degraded vegetation species (Nilsson and Berggren, 2000). The fragmentation results of the riparian and proximal hillslope also provided significant information about the geometry of vegetation patches.

The geomorphologic impacts of impoundments on the Keiskamma River were studied by Rowntree and Dollar (1994) and McGregor (1999). Rowntree and Dollar (1994) estimated that the Sandile Dam has a sediment trap efficiency of 100%. The 30% reduction in flood levels noted by Rowntree and Dollar (1994) due to the impoundment reduces the rates of river meandering and channel realignment, resulting in the narrowing of the riparian zone and a reduction in the patchiness and diversity of vegetation adjacent to the river channel (Stromberg, 1993). Decamp *et al*. (1988) point out that changes in the hydrological regime such as a reduction or elimination of the perturbing effects of floods and lowered groundwater levels introduce a new succession of riparian vegetation. Auble *et al*. (1997) also state that geomorphologic changes imposed by flow regulation may cause plant species to disappear. In the same vein, in the Keiskamma catchment, impoundments largely explain the increasing vegetation fragmentation within the riparian and proximal zones. Encroachment of degraded species such as blue bush (*Pteronia Incana*) and renosterbos (*Elytropappus rhinocerotis*) into the Keiskamma river valley is a further manifestation of the loss of saturation in the riparian zones and adjacent hillslopes. Well timed water releases from the Sandile and Binfield Park Dam are recommended in order to maintain vegetation in the riparian zone and proximal slopes.

As can be noted from Figure 3.12 and Figure 3.13, the different communal villages exhibit contrasting rangeland conditions. This contrast was also confirmed using landscape function analysis. A comparison of the rangeland conditions in the different communal areas confirms that significant differences exist in the mean landscape organisation index. Landscape function analyses indicate very low vegetation connectivity in highly degraded communal areas with weak local traditional institutions such as Zigodlo village. In contrast, communal villages with strong local governance institution such as Koloni are more pristine as shown by their much higher mean landscape organisation index. Bennett and Barrett (2007) investigated the grazing management system in the communal areas in the former Ciskei homelands and identified that the differences in the management systems are dependent on the degree of control the communities exert on communal grazing resources. Moyo *et al*. (2008) also concluded that rangeland condition and grazing strategies found in the communal areas are a sequence of the interaction between social, land tenure, ecological and institutional factors. Grazing resources are influenced by the social and ecological heterogeneity that characterise the catchment (Bennett and Barrett, 2007). Ainslie (2002) attributed the vegetation condition in the communal grazing areas to high stocking density and ineffective rangeland management methods. Difference in the strength of local institutions such as Residence Associations (RA) and traditional authorities responsible for coordinating grazing and land management in communal villages account for the variations observed in vegetation condition. The former commercial farms had the highest landscape index, reflecting relatively high vegetation connectivity although this was not significantly different from that of communal villages with good rangeland condition. The former commercial farms show higher vegetation connectivity due to very limited use compared to the communal areas.

Commercial farms however, showed higher variances in landscape organisation indices compared to communal areas due to increasing access to grazing in some areas, while areas not subjected to grazing remain in good condition. The differences in the current communal rangeland condition could be attributed to the role of local traditional institutions in controlling grazing lands. Bennett and Barrett (2007) also observed that co-existence of a statutory RA and an informal grazing committee weakens the influence of the RA in most former homelands. The dominance of the open-access system is a reflection of a weakening obligation to manage communal resources (Ainslie, 1998).

3.5 Conclusion

The study has confirmed that image objects in object oriented post-classification carry more information and are more suitable as ecological units for change detection purposes. The mixed pixel effect usually experienced in per pixel classification is minimized. The results of the study show increasing degradation trends in the Keiskamma catchment. A decrease in vegetated areas and bare patches has been noted between 1972 and 2006. The riparian and hillslope proximal zones also show evidence of fragmentation. This could be attributed to anthropogenic impacts as such as overgrazing, deforestation, cultivation and possibly permanent loss of saturation induced by river impoundments. The semi-arid communal areas in the central Keiskamma catchment showcase increasing degradation trends, particularly vegetation fragmentation. The differences in vegetation condition observed in the communal villages are a reflection of the interaction between social, land tenure, ecological heterogeneity and institutional factors that characterize the catchment. Vegetation condition is influenced by the strength or weakness of local institutions responsible for coordinating grazing and land management in communal areas. Degraded vegetation is more prevalent in villages with weak governing institutions and dilapidated infrastructure. In contrast, villages with strong traditional institutions which regulate grazing practices and enforce community rules still maintain healthy vegetation conditions. The overall results indicate that the environmental health status of the Keiskamma catchment is endangered by increasing degradation trends. A systematic re-allocation of state land in sections of the Keiskamma catchment which belonged to the former commercial farms is recommended. This will ease the current pressure on land

and enhance its judicious use. Properly regulated water releases from the Sandile and Binfield Park Dam are also recommended in order to maintain vegetation in the riparian zone and proximal slopes.

This chapter focused on assessing historical land use/cover changes and fragmentation processes in the catchment. Although this approach provided critical insights into land degradation processes occurring in the catchment, prediction of future land use/cover scenarios is an important requirement for rural land use planning. The next chapter focuses on simulating future land use/cover scenarios using the Markovian Cellular Automata model.

Chapter 4: Predicting future land cover changes in the Keiskamma catchment using a Markov-Cellular Automata model

4.1 Introduction

Land use/cover change dynamics is a central theme in global environmental degradation and sustainable land use planning (Lambin, 1999). The importance of land use/cover change dynamics in rural land use planning in sub-Saharan Africa has been highlighted by many studies (Lambin and Ehrlick, 1997; Reid *et al*., 2000; Lambin *et al*., 2003; Markland and Batello, 2008; Kamosoko *et al*., 2009). Projecting land cover changes and surface processes at regional scale is important in predicting areas that are susceptible to land degradation (Lambin *et al*., 1993). Modelling and simulating future land cover change provides an important means of assessing future land use/cover change and its relationship with driving forces (Lambin *et al*., 1993; Zhu *et al*., 2010). One motivation for modelling land use/cover change is to examine the effects of projecting short-term landscape dynamics over the long term (Urban and Wallin, 2002). Verburg *et al*. (2010) indicate that modelling is one of the most effective means of understanding trends in land use/cover change and in formulating effective land use policies. Simulation of likely future land use/cover scenarios aid planning support systems that are used in assessing alternative management scenarios (Torrens, 2006; Urban and Wallin, 2002). Modelling land use/cover changes also provides an understanding of the mechanisms underlying land use change (Huang *et al*., 2007), and is critical in the assessment of consequent environmental impacts (Houet and Hubert-Moy, 2006). Gómez-Mendoza *et al.* (2006) highlight that predictive models that focus on scenario simulation are essential in drafting sustainable development policies that preclude environmental degradation. Models of land use/cover change provide insights into the rate and spatial distribution of land use/cover change (Veldkamp and Lamb, 2001). Satellite remote sensing and GIS have been successfully utilised in the examination of land use and land cover change, particularly in quantifying the type, amount and location of land use/cover change (Wu *et al*., 2006). Technological advancements in remote sensing have ushered in new vistas in land use and land cover modelling through the provision of integrated software tools with geospatial, remote sensing and stochastic modelling capabilities (Weng, 2002; Zimmermann *et al*., 2007).

A variety of land cover change models have been developed for predicting landscape change at different levels of complexity (Veldkamp and Fresco 1996; Verburg *et al*., 1999; Lambin *et al*., 2000; Dietzel *et al*., 2005). Most techniques predict future scenarios based on the logistic regression, multi-agents and cellular automaton (Houet and Hubert-Moy, 2006). Prediction models can also be viewed as either stochastic or processes based. Examples of stochastic models include Markov, cellular automata, and logistic regression, while processes-based models include the dynamic ecosystem model (Lambin, 1994; Oğuz, 2004). These spatial models consist of three components: multitemporal maps, a transition function, and a simulated map of future land cover changes (Lambin, 1994; Jennerette and Wu, 2001; Attua and Fisher, 2010). The Markovian Cellular Automata model has gained standing as one of the most powerful means of projecting land use/cover trends (Petit *et al*., 2001; Houet and Hubert-Moy, 2006; Guo *et al*., 2009; Ye and Bai, 2008; Kamusoko *et al*., 2009; Attua and Fisher, 2010).

The Eastern Cape Province is one of the most degraded provinces in South Africa. Communal areas located in the former black homelands are the most degraded (Garland *et al*., 2000; Le Roux *et al*., 2007; Le Roux *et al*., 2008). The Keiskamma, located in the former Ciskei homelands is one of the catchments that is severely affected by soil erosion and thicket degradation (D'Huyvetter, 1985; Marker, 1988; Weaver, 1991). Many studies in land use/cover change using remote sensing in South Africa have been limited to change detection and identification of degraded land cover with little focus directed towards predicting future degradation scenarios. Quantitative information on land cover change that effectively describes prognostic land cover trends remains scarce. Efforts to effectively address land degradation have been constrained by the inadequate knowledge of land use/cover dynamics. Scoging and Lent (2000) point out the need for predictive methods in natural resource monitoring in the Eastern Cape, South Africa, which will enable environmentalists to get crucial insights into future environmental threats. Thus, the objective of this study is to simulate and predict the future land use/cover changes in Keiskamma catchment using the Markovian Cellular Automata model.

4.2 Markov Cellular Automaton modelling of land use/cover changes

A Markov-cellular automaton is a robust spatially explicit hybrid model which integrates the Markov Chain Analysis and Cellular automata and is an improvement in spatio-temporal dynamic modelling (Silvertown *et al*., 1992; Li and Reynolds, 1997; Wu and Webster, 1998; Houet and Hubert-Moy, 2006). The integration of the Markov process and the cellular automaton mechanism offers significant modelling advantages. Whereas the Markov process directs temporal dynamics among the land cover classes by means of transition probabilities (Turner, 1987; Silverton *et al*., 1992; Jennerette and Wu, 2001) the cellular automaton mechanism addresses the local rules relating to neighbourhood configuration. In tandem with the transition probability, it determines the spatial dynamics of land cover types (Silvertown *et al*., 1992; Wu and Webster, 1998; Houet and Hubert-Moy, 2006). Although the Markov chain analysis operates under fairly restrictive assumptions such as independence and stationarity, it is mathematically easy to implement (Wood *et al*., 1997). In addition, the land use/cover transition probability results can serve as an indicator of the direction of land use/cover processes (Weng, 2002). Deficits in the Markov Analysis are compensated through the integration of the cellular automata, which facilitates the spatial interactions of the land covers through proximity modelling (Clarke *et al*., 1997; Houet and Hubert-Moy, 2006; Ye and Bai, 2008).

4.3 Data and methodology

The datasets used for predicting the land use/cover changes include land use/cover maps previously generated by object oriented classification in Chapter 3 using Landat 5 TM satellite imagery. The 1993 and 2006 classified maps were used for the projection to 2019. 1992, 1997 and 2001 land use/cover maps were used for validation purposes. The 1972 and 1988 land use/cover maps together with above mentioned datasets were used to determine the vegetation trends in the catchment, for suitability analysis. The overall accuracies achieved in the classification maps used in this study are 0.861, 0.819, 0.893, 0.899, 0.920, and 0.898 for 1972, 1988, 1997, 2001, 2006 respectively. Shapefiles for areas designated for settlement were extracted from the Amatole District Municipality's Land Reform and Settlement Plan (2007). Idrisi Andes software was used for performing the Markovian Cellular Automata model and model validation. This remote sensing and GIS software was selected for its advanced environmental modeling capability.

4.3.1 Markovian simulation

The Markovian Cellular Automata model was implemented to predict land use/cover changes in this study. This model was chosen based on its simplicity to implement in a GIS environment. Markov Chain Analysis is suitable to use when changes and processes in the landscape are complex to describe. A Markov process is defined as one in which the future state of a system is projected entirely on the basis of the immediately preceding state. The process involves computing the transition probability matrix of land cover change from time one to time two, which is then considered to be the basis upon which to assign to a later time period. In this study, the Markov chain method was implemented to analyse the 1993 and 2006 pair of classified images and to generate a transition probability matrix, a transition area matrix, and a set of conditional probability maps. A transition probability matrix indicates the probability of inter-class transitions among different land use/cover types, while a transition area matrix shows the quantity of land that is expected to transform from one class to another over a 13 year period (up to 2019). Conditional probability images show the probability of existence of a particular land use/cover type over the 13 year period; these images are computed as temporal projections based on the 1993 and 2006 input land use/cover images. The 1993 and 2006 classified maps were used as the earlier and later land cover images respectively. The prediction is purely based on the state of land cover in 1993 and 2006; the background cells were assigned a value of 0.0. A proportional error of 0.11 was assigned to the prediction based on an overall accuracy of 89% for 2006. Land use/cover is considered to be temporally persistent over 10-15 year intervals (Lambin *et al*., 1999; Gómez-Mendoza *et al*., 2006), thus a 13 year prediction used in this study was within the required range.

A summary of the computations involved in the Markov projections is shown below, where land use/cover is considered as stochastic process of which the different classes are regarded as the states of a chain (Weng, 2002).

A Markov chain is expressed as follows:

$$P(X_t = j | X_o = i_o, X_1 = i_1, ..., X_{t-1} = i_{t-1})$$
$$= P(X_t = j | X_{t-1} = i_{t-1}). \qquad (1)$$

If a Markov sequence of random variable X_n takes the discrete values $a_1 .... a_N$, then

$$P(x_n = a_{i_n} | x_{n-1} = a_{i_{n-1}}, ..., x_1 = a_{i_1})$$
$$= P(x_n = a_{i_n} | x_{n-1} = a_{i_{n-1}}) \qquad (2)$$

Where the sequence x_n is called a Markov chain.

The initial transition area matrix of the different land use/cover classes is shown in expression (3).

$$X_{ij} = \begin{bmatrix} X_{11} & X_{12} & ... & X_{1n} \\ X_{21} & X_{22} & ... & X_{2n} \\ ... & ... & ... & ... \\ X_{n1} & X_{n2} & ... & X_{nn} \end{bmatrix} \qquad (3)$$

Where X_{ij} indicates the quantity of land use/cover type *i* transforming to land use/cover *j* over a particular period of time, *n* denotes the number of land use/cover types. This calculation simplifies to:

$$\sum_{j=1}^{n} X_{ij} = X_i, \ \sum_{i=1}^{n} X_{ij} = X_j \qquad (4)$$

$$P_{ij} = X_{ij} / X_i \qquad (5)$$

P_{ij} shows the ratio of the quantity of land use/cover type *i* transforming into the land use/cover class *j* in the period of time. Using the equations of the Markov process and Bayesian principles of conditional probability, the above equations are further simplified to.

$$\pi_j(k) = \sum_{i=1}^{n} \pi_i(k-1) P_{ij} (j = 1,2,...,n) \qquad (6)$$

Where $\pi_j(k)$ denotes the area of land use type j at the *k*th state.

4.3.2 Transition rules: Land use/cover suitability criteria

Land cover suitability images were derived to determine the transition suitability of each pixel for each land use/cover type. The suitability criteria for vegetation, degraded vegetation, bare and degraded soil and water was based on temporal analysis of land cover trends from 1972 to 2006. A similar technique was used by Ye and Bai (2008) to derive suitability images. The state-and-transition model used in rangeland ecology was used to understand the processes underlying land cover change dynamics (Briske *et al*., 2005). These principles were applied to determine suitable sites for intact vegetation and degraded vegetation, bare and degraded soil because state-and-transition models accommodate greater complexity by considering vegetation dynamics in response to multiple drivers and by characterizing transitions to alternative stable states on individual ecological sites (Briske *et al*., 2005). Vegetation dynamics are characterized by continuous reversible and discontinuous non-reversible trends (Wu and Loucks, 1995; Watson *et al*., 1996; Illius and O'Connor, 1999). The occurrence of continuous and reversible vegetation dynamics is dominant in stable vegetation states. Discontinuous and non-reversible dynamics result once one stable state replaces another, when thresholds have been exceeded. Ecological thresholds are difficult to identify since ecosystem modification often imposes a series of feedback mechanisms that maintain or reinforce the altered state and limits reversal to the previous stable state (Archer *et al*., 2001; Scheffer *et al*., 2001; van de Koppel *et al*., 2002). It is noteworthy however, that vegetation dynamics exhibit complex trends difficult to model without simplifications. Given that predictive vegetation mapping is based on the ecological niche theory and gradient analysis. This study therefore assumes that suitable sites for vegetation are ecological niches in which vegetation established itself in the past when anthropogenic effects were minimal and climatic factors favourable. The distribution of settlements in the Keiskamma catchment is characterised by a mixture of land tenure systems that exist in the region (Ruhiiga, 2000; Bank and Minkley, 2005). Such complexities are difficult to model without simplifications.

The following procedure was thus followed to derive suitable sites for the different land cover types, suitable sites were assigned a weight of 1 and none suitable sites a weight of 0.

(1) Suitability sites for vegetation were derived by combining the 2006 vegetation and degraded vegetation sites; this logic permits the transformation of vegetation towards recovery or further degradation.

(2) Degraded vegetation follows a similar logic applied to intact vegetation. The 2006 degraded vegetation class was combined with 2006 vegetation and bare and degraded soil class.

(3) The suitability criteria for bare and degraded soil were achieved by combining the 2006 bare and degraded soil class with the degraded vegetation class. Such suitability criteria permit further deterioration of degraded vegetation into bare and degraded soil.

(4) Water suitability sites were derived from the 1993 land cover map; this showed the highest sites of water features from 1972 to 2006.

(5) Suitability sites for settlements were delineated from a digital shapefile outlining areas designated for settlements in the catchment. This built-up layer shows the major boundaries of communal villages, towns and other planned settlements. The shapefile was rasterized and all current and planned settlements assigned a suitability of 1 and non-settlements were allocated a value of 0. This procedure conforms to LRSP's strategy for densification and formalization of existing settlement and its new sites for planned settlement.

A fuzzy set membership function was used to standardize the values by converting the binary images into byte data format (0-255), the suitability maps will indicate the transition probability of each pixel to fit into a specific land use/cover class. A transition suitability image collection was created using a set of the suitability images.

4.3.3 Integration of the Markov Chain analysis and Cellular Automata

Markov Chain analysis results were further processed using the Markov Cellular Automata algorithm to bring a spatial sense not considered in the Markov Chain projection (Houet and Hubert-Moy, 2006). The Markov Cellular Automata function integrates the Cellular Automata, Markov Chain and Multi-Objective Land Allocation which takes consideration of spatial contiguity and a sense of the likely spatial distribution of the transitions to Markov chain analysis (Eastman, 2006). The 2006 land cover classification was used as a basis land cover image for change simulation. A transition area file derived from the Markov Chain analysis was incorporated into the Markov Cellular Automata computation, which determines the quantity of potential land allocated to each land cover class over a 13 year period. A 5x5 contiguity filter shown in equation (7) was chosen for the cellular automata. The filter down-weights the suitability of pixels that are far from existing areas of each land cover class.

$$\begin{bmatrix} 0 & 0 & 1 & 0 & 0 \\ 0 & 1 & 1 & 1 & 0 \\ 1 & 1 & 1 & 1 & 1 \\ 0 & 1 & 1 & 1 & 0 \\ 0 & 0 & 1 & 0 & 0 \end{bmatrix} \qquad (7)$$

The role of the contiguity filter is to ensure the ideal choices for land cover transformation are restricted to cells that are both inherently suitable and in close proximity to existing areas of that land cover class; this gives preference to contiguous suitable areas. A total of 13 iterations were used in the simulation. The multi-objective land allocation (MOLA) procedure was used in each time step to resolve the land allocation conflicts. All land use/cover classes act as claimant classes and contend for land within the host class (Eastman, 2006).

4.3.4 Model Validation

The validity of the Markovian Cellular Automata simulation was assessed using advanced multi-resolution statistical algorithms proposed by Pontius (2002) to measure the agreement between two categorical images. Agreements between a pair of maps are assessed in terms of location and quantity of cells in each category by computing various Kappa Indices of Agreement and related statistics. In this study, 1993 and 1997 classified maps were projected to 2001; a comparison was then made between a 2001 simulated map and a reference map for 2001 produced from a satellite-derived classification map.

4.4 Results

4.4.1 Predicted land cover transformations from 2006 to 2019

The land use/cover maps used in the projection are shown in Figure 4.1 (1993) and Figure 4.2 (2006) while the simulated land use/cover map for 2019 is shown in Figure 4.3. Transition probabilities and areas tables are shown in Table 4.1 and Table 4.2. The transition probabilities indicate a probability of 0.4474 for vegetation to remain in its current state and a probability of 0.5132 for vegetation to transform to degraded vegetation. The conversion of vegetation to degraded vegetation state is accompanied by an area migration of 455.036km^2. A lower probability of 0.0195 is associated with direct vegetation migration to bare and degraded soil, accompanied by a transition area of 17.262 km^2.

A probability of 0.0488 and a transition area of 74.179 km^2 is associated with a further degradation of degraded vegetation into bare and degraded soil. A higher probability of 0.4889 exists that bare and degraded soil will recover to degraded vegetation with an area coverage of 35.222km^2. Probabilities of 0.2343 and 0.179 exist for bare and degraded soils and degraded vegetation respectively to recover to fully vegetated areas with transition areas of 16.884 km^2 and 271.991 km^2 respectively associated with the recovery. The high transition probability from water to vegetation is mainly as result of the reduction in flow regimes in the Keiskamma River. Geomorphic processes such as channel narrowing due to the impoundments of the Keiskamma River could explain the high transition from water to vegetation. The 2019 simulated land use/cover maps reveal significant narrowing of the riparian zone. A change detection comparison of the 2006 classification and 2019 simulation image was undertaken to determine the patterns of land cover/use transformations to be expected in 2019. The change detection matrix statistics in Table 4.3 reveal that 23.576% of the vegetation cover will transform to degraded vegetation, whilst a mere 3.217% of degraded vegetation is going to recover to full vegetation cover. A further 8.147% of degraded vegetation will degrade to bare and degraded soil whilst only 3.62% of bare and degrade soil will transform to degraded vegetation. The transformation from bare and degraded soil to degraded soil might however not signal recovery, but encroachment by alien invasive species such *Pteronia incana* whose inferior patchy vegetation cover tends to promote soil erosion.

Table 4.1 Land use/cover transition probabilities, 2006-2019.

2006:	Probability of changing: 2019:				
	V	W	DV	BDS	S
V	0.4474	0.0003	0.5132	0.0195	0.0196
W	0.3761	0.1213	0.4423	0.0133	0.047
DV	0.179	0.0004	0.612	0.0488	0.1598
BDS	0.2343	0.0046	0.4889	0.0283	0.2438
S	0.1912	0	0.4754	0.1487	0.1847

Table 4.2 Land use transition area matrix (in km2) 2006-2019.

2006:	Expected transition: 2019				
	V	W	DV	BDS	S
V	396.7236	0.2556	455.0364	17.262	17.3781
W	2.3292	0.7515	2.7405	0.0828	0.2916
DV	271.9908	0.5760	929.6919	74.1789	242.7543
BDS	16.884	0.3348	35.2224	2.0421	17.5653
S	39.9834	0	99.4239	31.1067	38.6181

A net decrease of 17.993% and 0.116% was recorded for vegetation and water respectively. Significant increases of 78.46% and 52.841% were projected for settlement, bare and degraded soil whilst a marginal net growth of 0.0115% for degraded vegetation is predicted. The changes which will occur in terms of area are shown on the clustered column graph (Figure 4.4). The graph clearly illustrates the declines in vegetation and an increase in settlements, bare and degraded soils. The change detection statistics confirm considerable degradation to bare and degraded soil and conversion of vegetation to settlements. Although a minor net change in degraded vegetation is predicted, examination of the change detection matrix reveals that important land use/cover class interchanges are concealed by viewing the net changes per land use/cover class alone. The changes in degraded vegetation are characterized by significant losses to bare and degraded soils which are compensated by gains from intact vegetation; such a scenario indicates an increase in land degradation.

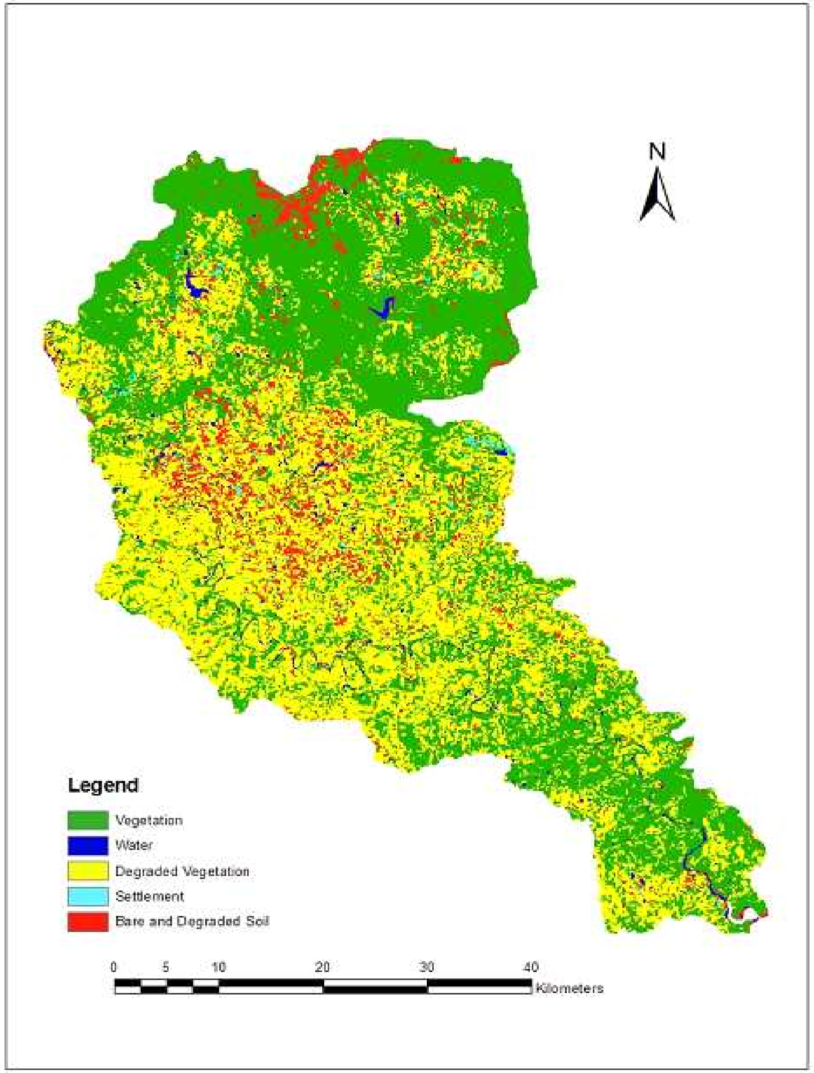


Figure 4.1 1993 LULC Classification

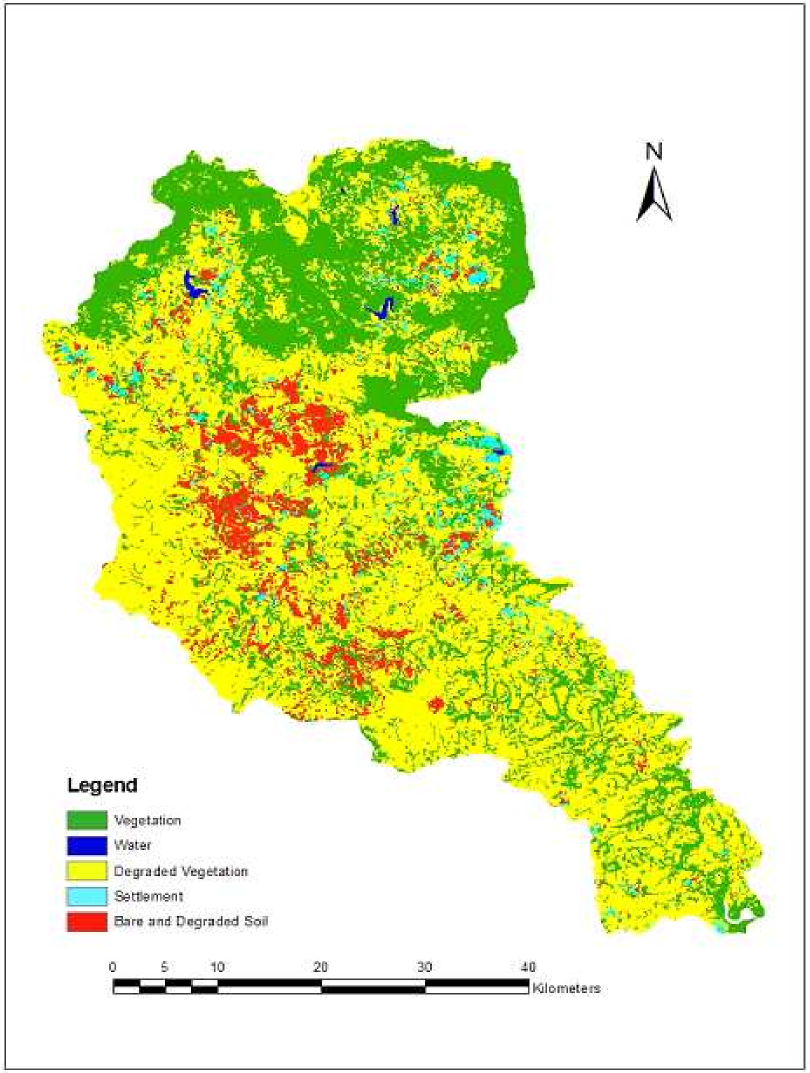


Figure 4.2 2006 LULC Classification.

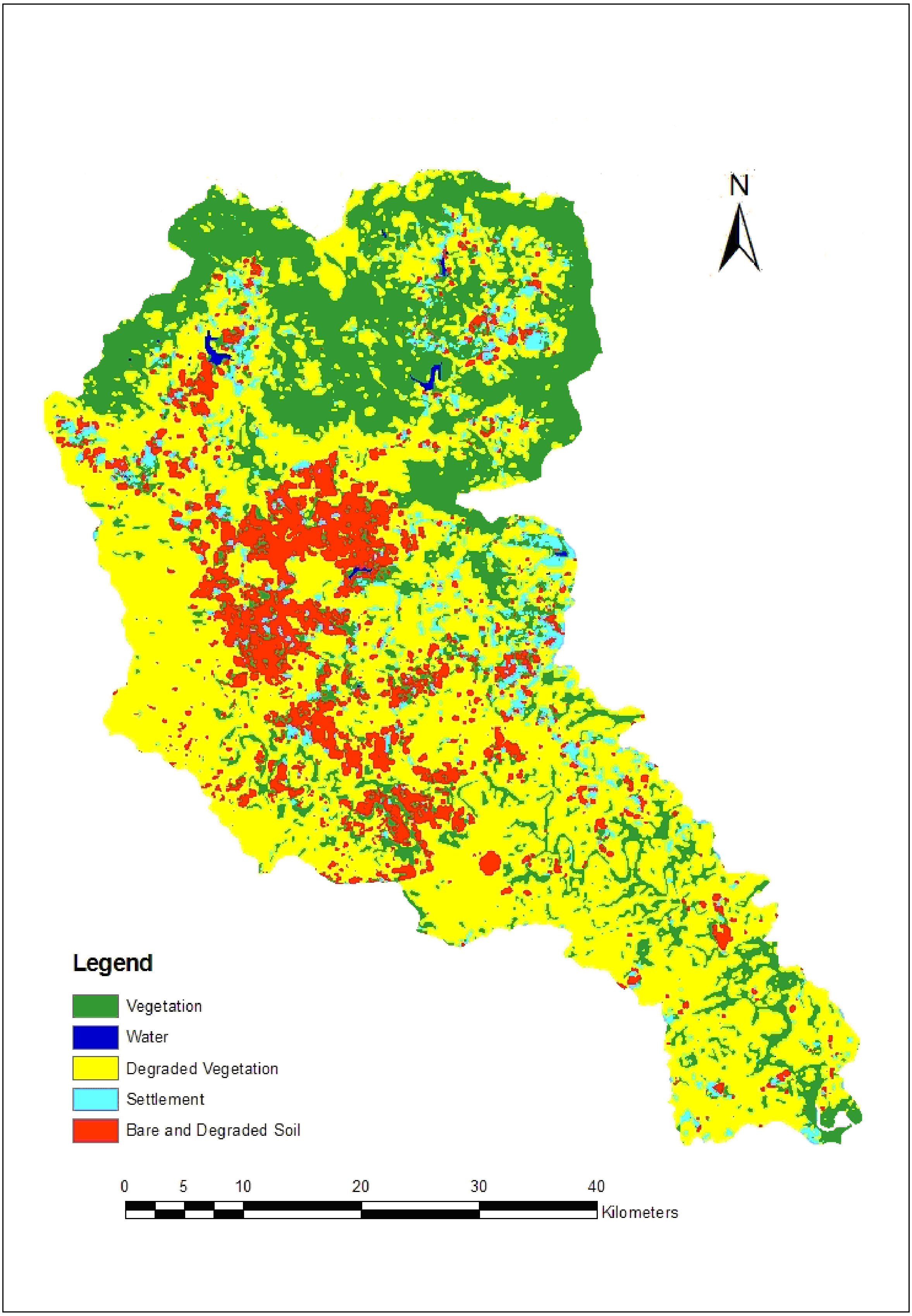


Figure 4.3 Simulated land use/cover classes for 2019.

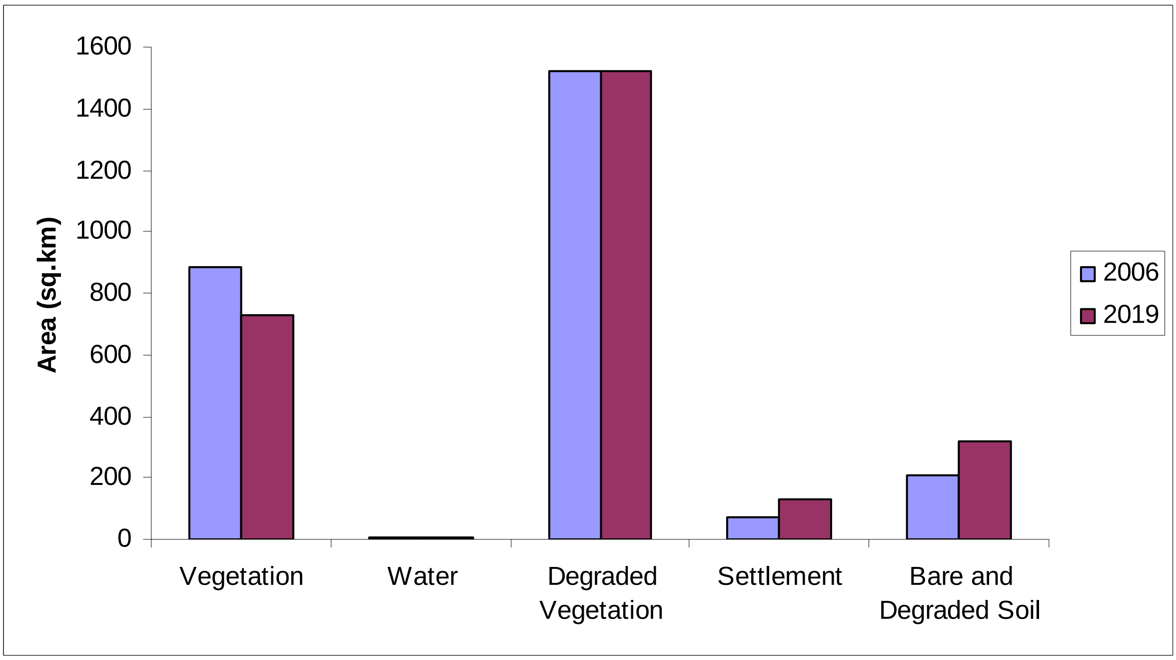


Figure 4.4 Comparison of the 2006 LULC classes with the 2019 simulated LULC.

Table 4.3 Change detection matrix (in %) 2006-2019.

Final State:2019	Initial State: 2006						
	V	W	DV	S	BDS	Row Total	Class Total
V	75.977	0.015	3.217	0.047	0.001	99.373	100
W	0.004	95.467	0.002	0.266	0.008	100	100
DV	23.576	0.029	85.611	0.024	3.62	99.755	100
S	0	0	3.024	96.406	5.842	99.25	100
BDS	0.443	4.489	8.147	3.257	90.529	100	100
Class Total	100	100	100	100	100	0	0
Class Changes	24.023	4.533	14.389	3.594	9.471	0	0
Image Difference	-17.993	-0.116	0.115	78.46	52.841	0	0

4.4.2 Model Validation

Validating future land use/cover scenarios is generally a problematic (Houet and Hubert-Moy, 2006) but necessary procedure. The model validation results indicate that the 2001 projected results are in good agreement with the 2001 reference map. This deduction is based on the high indices of agreement presented in Table 4.4 and Table 4.5. For instance a Kappa Index of Agreement (KIA) of 0.7505 shows very good agreement, whilst a KIA of 1 reflects a perfect match between the simulated map and the reference reality map. The KIA for grid-cell level location is 0.8299, reflecting that grid cells are well located in the landscape. A KIA for stratum-level location of 1 reflects that the grid cells are perfectly located within the strata. Very strong agreements between the simulated and reference maps are shown at medium and perfect information levels of quantity. A comparison of the simulated map for 2001 and the classified map are shown below on Figure 4.5. Houet and Hubert-Moy (2006) point out that over-estimations and under-estimations are likely to occur when one uses short-term trends. This could be the case for a 4 year period used in the validation. It is envisaged that longer time periods such as 13 years used in the 2006 to 2019 simulation produce more improved results. The validation process shows that the Markov Cellular Automata prediction has got a high chance of predicting the future scenarios based on the KIA. It can therefore be concluded that the Markov Cellular Automata is a feasible means to predict future land use/cover states and is a useful tool to assist environmental planning.

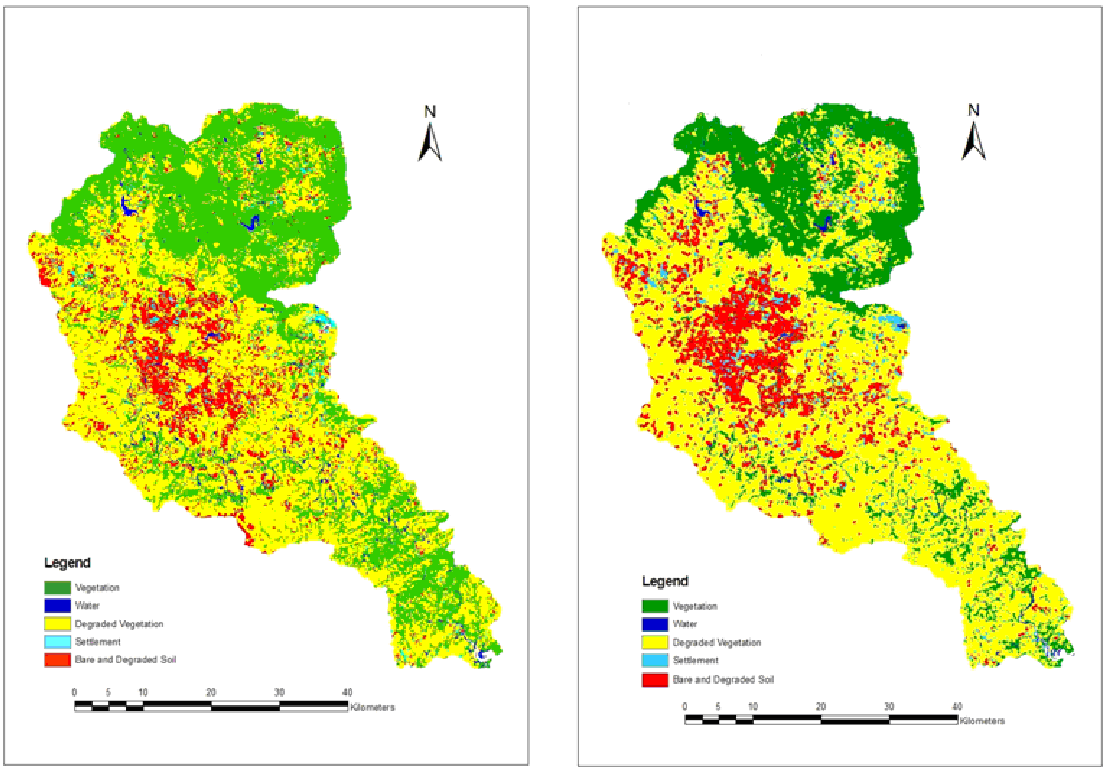


Figure 4.5 Comparison of the 2001 classified LULC (left) and simulated LULC (right).

Table 4.4 Categorical image comparison of 2001 classification and 2001 projection.

Classification agreement/disagreement According to ability to specify accurately quantity and location		*Information of Quantity*	
Information of Location	No[n]	Medium[m]	Perfect[p]
Perfect[P(x)]	P(n)=0.486	P(m)=0.940	P(p)=1
PerfectStratum[K(x)]	K(n)=0.396	K(m)=0.845	K(p)=0.839
MediumGrid[M(x)]	M(n)=0.396	M(m)=0.845	M(p)=0.838
Medium Stratum[H(x)]	H(n)=0.396	H(m)=0.845	H(p)=0.839
No[N(x)]	N(n)=0.143	N(m)=0.377	N(p)=0.378

Table 4.5 Validation summary of agreements and kappa indices.

Agreement due to chance	0.143
Agreement due to quantity	0.235
Agreement due to location at the stratified level	0.467
Agreement due to location at the grid cell level	0.000
Disagreement due to location at the grid cell level	0.000
Disagreement due to location at the stratified level	0.096
Disagreement due to quantity	0.060
Kappa for no information	0.819
Kappa for grid-cell level location	0.830
Kappa for stratum-level location	1.000
Kappa Index of Agreement (Kappa Standard)	0.751

4.5 Discussion

The Markov Cellular Automata simulation results predict further reductions in intact vegetation and subsequent increases in bare and degraded soils. Such a scenario suggests land degradation will continue if the current trends are persistent. Kamusoko *et al.* (2009) indicate that future declines in vegetation and increases in bare land have severe implications for degradation and impacts negatively on rural livelihoods. Whereas the simulations predict a marginal net increase in degraded vegetation, the transformations are characterised by important land use/cover class interchanges, the area losses incurred due to degradation to bare and degraded soil are compensated by some gains accrued from intact vegetation. Muller and Middleton (1994) caution that minor variances in the total amounts of land conceal important interchanges when simple land use distributions are examined. In this simulation degraded vegetation seems to be acting as a catalyst through a series of feedback mechanisms in the interaction of intact resident vegetation and degraded bare soils and transforms the catchment to degradedness. The function of degraded vegetation in accelerating degradation in ecosystems has been observed in previous studies (Kakembo, 2009). The simulation results also indicate significant narrowing of the riparian zone. This has important implications to ecosystem function if the current trends persist. Noteworthy is the high transition probability from water to vegetation, this could be attributed to a reduction in flow regimes and channel narrowing. This trend is consistent with the observation made by Rowntree and Dollar (1994) and was attributed to the long term impacts of impoundments. The results also indicate increases in settlement, this could be a result of natural population growth. The predicted scenario can however be prevented if effective environmental strategies to curb deforestation and overgrazing are put in place. An integrated environmental policy to curb degradation should be developed with contributions from the communities. The imposition of environmental legislation into communities without sufficient community consultation in policy formulation has yielded poor results in the past (Bennett and Barrett, 2007; Moyo *et al*., 2008). The betterment programme implemented in the catchment in the past is a typical example (De Wet, 1989). Effective environmental monitoring could be achieved if the local communities are educated to exploit their resources sustainably and become responsible custodians of their environment. This study provides an important contribution to environmental planning in South Africa.

The use of land use plans from local municipalities, temporal land cover trends from satellite imagery and theoretical state and transition models underlying land cover change used in this study proved to be a viable means for deriving suitability images. This criterion is comparable to a methodology proposed by Verburg *et al*. (2004) for deriving transition rules using theories underlying land use structure and quantification of neighbourhood characteristics using observed trends. Petit *et al*. (2001) point out that it is critical to make certain restrictive assumptions to project future land cover changes. Setting up transition rules for Markov Cellular Automata modelling is usually problematic (Verburg *et al*., 2004). This study thus provides a simple approach and contributes to modelling research in rural landscapes. The study also demonstrates the feasibility of using the Markov Cellular

Automata for projection of land cover trends regardless of whether the trends persist or not, this has been corroborated in many studies (Gómez-Mendoza *et al*., 2006; Wu *et al*., 2006; Kamusoko *et al*., 2009; Attua and Fisher, 2010). Furthermore, the descriptive power of the Markovian model was useful in explaining the land cover trends, a similar assertion was made by many scholars (Petit *et al*., 2001; Weng *et al*., 2002; Wu *et al*., 2006; Kamusoko *et al*., 2009). Land use/cover is considered to be temporally persistent over 10-15 year intervals (Lambin *et al*., 1999; Gómez-Mendoza *et al*., 2006), thus a 13 year prediction done in this study provides useful insights for future scenarios. One limitation of Markovian models in land cover projections is that the influence of exogenous and endogenous variables to the transitions cannot be incorporated into the models in order to better understand land use and land cover processes (Weng, 2002). For instance the effect of climate change and other extreme climatic events such as drought, floods, excessive rain and unforeseen political interventions has not been considered in this study, this has considerable potential to affect the predicted scenarios. That notwithstanding, the Markov Cellular Automata model still provides useful future scenarios for planning purposes and are widely used as confirmed by previous studies (Wood *et al*., 1997; Petit *et al*., 2001; Weng, 2002; Gómez-Mendoza *et al*., 2006; Ye and Bai, 2008; Guo *et al*., 2009; Kamusoko *et al*., 2009). The validation results also indicate that the Markov Cellular Automata simulation provides reasonable results for planning purposes.

4.6 Conclusion

It has been demonstrated in the present study that the Markov Cellular Automata is a feasible modelling method to predict future land cover/ use states and should be integrated into environmental planning processes. Simulation results reveal declines in vegetation cover in 2019. The predictions also suggest significant increases in bare and degraded soil and human settlement in 2019. No significant changes were predicted for degraded vegetation, however areas lost as a result of transformation to bare and degraded soil are compensated by gains claimed from intact vegetation. Such a scenario indicates an increase in land degradation. The simulation suggests significant narrowing of the riparian zone and a high transition from water to vegetation. If the current trends persist the simulated scenario will have adverse effects on ecosystem function. This study provides a useful prediction that could serve as an early indicator of possible future scenarios if the current land cover trends persist. It also demonstrates that a predictive and proactive approach to environmental degradation can be adopted rather than a reactive approach still prevalent in Southern Africa. The simulation results can however be prevented if effective environmental strategies to curb land degradation are put in place.

Land degradation is controlled by a number of variables which include injurious land use, vegetation cover, climatic factors, soil erodibility, conservation practice and topographic parameters. Chapter 3 and Chapter 4 focussed on modelling land use/cover changes, which are important factors controlling land degradation. Land degradation is also in most cases manifested through soil erosion. An assessment of soil erosion risk is thus required to provide a holistic overview of the soil loss patterns and soil erosion potential of the catchment. The next chapter integrates the main factors affecting soil erosion to determine the soil loss patterns, the sediment transfer processes and the physico-chemical characteristics of the soils.

Chapter 5: Soil Erosion Risk Assessment of the Keiskamma Catchment using GIS and Remote Sensing

5.1 Introduction

Soil erosion by water is a major environmental problem that threatens the ecological function of terrestrial and aquatic systems worldwide (Oldeman, 1994; Nyakatawa *et al*., 2001). It is estimated that 85% of global land degradation is associated with soil erosion and close to 5Mg ha^{-1} per year of productive topsoil is lost to lakes and oceans in Africa (Oldeman *et al*., 1990; Angima *et al*., 2003). Flügel *et al*. (2003) predict that soil erosion will become more severe in Southern Africa due to population increases and climatic changes. More than 70% of South Africa is affected by soil erosion of varying intensities (Garland *et al*., 2000; Le Roux *et al*., 2008). Le Roux *et al*. (2008) highlight that the Eastern Cape Province has one of the highest erosion potentials in South Africa.

Soil erosion is a natural process and relates to the entrainment and transportation of earth materials across a given surface. Soil loss is defined as the amount of material that is actually removed from a particular slope (Renard *et al*., 1997), and is one of the major indicators of environmental degradation. The negative effects caused by soil erosion on soil degradation, hydrological systems, agriculture, water quality and the environment in general have long been established and the impacts of soil erosion continue to pose severe threats to human sustenance (Lal, 1998). The impacts of soil erosion include loss of fertile topsoil, decline of soil productivity and reduction in water quality in river networks. Reservoir sedimentation is one of the direct impacts of soil erosion that exacerbates water management problems in Southern Africa (Flügel *et al*., 2003). The economic and environmental impacts of accelerated soil erosion are difficult to quantify because of its extent, magnitude, rate and complexity of the processes related to it (Lal, 1994).

Timely and accurate estimation of soil loss or evaluation of soil erosion risk is now regarded as an issue of high priority. Many models have been developed to estimate soil loss (Wischmeier and Smith, 1978; Nearing *et al*., 1989; Adinarayana *et al*., 1994; D'Ambrosio *et al*., 2001; Veihe *et al*., 2001; Shen *et al*., 2003; Lim *et al*., 2005) and among them, the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978), Soil and Water Assessment Tool (SWAT) (Arnold *et al*., 1998), Soil Erosion Model (EUROSEM) (Morgan *et al*., 1998) and Water Erosion Prediction Project (WEPP) (Flanagan and Nearing, 1995) have been widely used. The USLE has been used successfully to estimate soil erosion potential for nearly 40 years (Dennis and Rorke, 1999; Kinnell, 2000). Process-based erosion models have limited use due to intensive data and computation requirements. RUSLE was developed on the basis of substantial modifications of the USLE and its database to more accurately estimate Rainfall erosivity (R), Soil Erodibility (K), Land cover management (C), conservation practice factor (P) factors, and soil erosion (Renard et al. 1991). The RUSLE includes the analysis of data that was not previously included in the USLE, and an update of the theory describing hydrologic and erosion processes. Renard *et al*. (1994) provide a detailed summary of the differences between USLE and RUSLE. Prominent modifications include corrections of the rainfall erosivity factor (R), new equations based on the ratio of rill to interrill erosion that accommodate complex aspects of slope length (LS) and the implementation of new subfactors for calculating the land cover management factor (C) and the new conservation practice factor (P) (Renard *et al*., 1991; Renard *et al*., 1994). Notwithstanding these modifications, the RUSLE model has retained the same fundamental structure as the USLE (Renard *et al*., 1994).

The RUSLE model has been used extensively in predicting soil loss around the world. Wischmeier and Smith (1978) originally developed the USLE for soil erosion estimation in croplands on gently sloping topography. While the RUSLE model has gained acceptance for use at river catchment and regional scales (Millward and Mersey, 1999; Bogg *et al*., 2001; Shi *et al*., 2004; Fu *et al*., 2005; Onori *et al*., 2006; Le Roux *et al*., 2008), USLE and RUSLE were initially developed to estimate soil erosion at small hillslope and plot scale (Le Roux *et al.*, 2007; Wischmeier and Smith, 1978). The SATEEC model can be used for soil erosion risk assessment at watershed scale because of the Sediment Delivery Ratio (SDR) module integrated in it (Lim *et al*., 2005; Park *et al*., 2010). The SDR is defined as the ratio of sediment yield to the total surface erosion as affected by catchment topography, land cover, sediment sources, transport system and texture of eroded material (Walling, 1988; Bhattarai and Dutta, 2007). The concept of SDR encapsulated within the RUSLE based SATEEC model is an important paradigm at catchment scale since significant sediment deposition occurs within the catchment before it reaches the catchment outlet (Bhattarai and Dutta, 2007). The SATEEC model is thus a substantial improvement of the RUSLE model since it incorporates spatially disturbed sediment delivery ratios to compute soil loss from rill and interrill erosion.

Mapping soil erosion in large areas using traditional methods is a difficult task. The use of remote sensing in conjunction with GIS techniques makes soil erosion estimation and its spatial distribution attainable at a higher accuracy and lower cost (Millward and Mersey, 1999; Wang *et al.*, 2003). The integrated application of remote sensing, GIS and RUSLE provides the potential to estimate soil erosion loss on a cell-by-cell basis (Millward and Mersey, 1999). Soil erosion risk was assessed successfully using RUSLE by Boggs *et al*. (2001) using a digital elevation model (DEM) and land-unit maps. Related studies also successfully applied the RUSLE model to soil erosion risk mapping using remote sensing and GIS techniques (Wang *et al*., 2003; Bartsch *et al*., 2002; Millward and Mersey, 1999; Reusing and Ammer, 2000; Ma *et al*., 2003; Boggs *et al*., 2001; Cerri *et al*., 2001; Bartsch *et al*., 2002). The RUSLE parameters can be altered significantly by human activities. The C factor can be changed by deforestation; the P factor can be transformed by shifting community environmental practices and the L factor by changing the dimensions of the fields.

Gullies are the dominant sources of sediment in Keiskamma catchment. Gully formation is often triggered by topographic parameters due to the physiographic influence on subsurface water movement, surface runoff, surface saturation zones and soil water distribution and soil water flux (Moore *et al*., 1988b; Moore and Burch, 1986; O'Loughlin, 1986; Zaslavsky and Sinai, 1981; Beven and Kirkby, 1979; Thorne *et al*., 1986). Ephemeral gully development is influenced by topographic variables such as surface saturation and stream transport capacity. Lentz *et al*. (1993) indicate that the most valuable topographic indices are planform curvature, profile curvature, slope, unit area/ slope, unit area, and upstream contributing area. The role of topographic thresholds in gully development is highlighted by many scholars (Moore *et al*., 1988b; Poesen, 2002; Vandekerckhove *et al*., 2000; Kakembo *et al*., 2009). Thorne *et al*. (1986) suggest gully initiation and expansion is also influenced by stream power, a parameter which is also topographically controlled. Thompson and Moore (1996) established a significant correlation between the topographic wetness index and the water table. Mahalanobis distance is a valuable method of measuring how similar some set of conditions are to an ideal set of conditions and this can be used to identify landscape zones that are most similar to some "ideal" landscape (Clark *et al*., 1993; Dettmer *et al*., 2002; Jenness, 2003). This method is ideal for discerning typical areas susceptible to gullying.

Physical and chemical properties of the parent materials and soils within sediment source areas provide critical information relating to the susceptibility of the soils to erosion by water (Jones, 2010; Verachtert *et al.*, 2010). Piping is also a key process in the transfer of sediment from hillslopes to river channels in the Keiskamma catchment. While empirical soil erosion models are an important means to evaluate soil erosion at various scales, processes such as crusting, piping and subsurface seepage cannot be completely accounted for in empirical soil erosion models. The USLE based models for instance, are designed to compute sheet (interrill) and rill erosion and are not well suited to simulate gully erosion and associated processes such as piping (Lim *et al*., 2005). Thus, soil erosion models need to be supported by physical and chemical analyses of field soil samples to gain an understanding of soil properties promoting soil erosion (Laker, 2004). Field observations, measurements and laboratory soil analyses can also be used as a means to calibrate and validate empirical models. The soil erodibility factor required for the USLE for instance, requires among other things, particle size analysis to establish the textural properties of the soils. Physico-chemical soil analysis is needed to explain critical aspects affecting soil erosion such as soil erodibility, crusting and piping, subsurface erosion and even the spatial distribution of soil erosion (Jone, 2010). Zhang *et al*. (2006) established relationships between soil erosion and some soil chemical property patterns. The role of soil chemistry and other intrinsic soil properties in the development of piping and tunnelling has been established (Qadir and Schubert, 2002). Chemical properties that strongly influence soil dispersivity such as exchangeable sodium percentage (ESP) and sodium adsorption ratio (SAR) are often not integrated into the USLE soil erodibility calculation. A physical and chemical analysis of the soil properties thus provides critical insights into the susceptibility of the soil to erosion (De Santis *et al*, 2010; Verachtert *et al.*, 2010).

Sediment analysis provides an integrated view of sediment sources, transfers, sinks and outputs of a drainage basin, and draws together the many aspects of erosion, sediment mobilization, transport, storage and yield. According to a review by Le Roux *et al*. (2007) of erosion assessment projects conducted in South Africa, the evaluation of soil erosion risk within the context of environmental degradation has not attracted sufficient scientific attention in the Eastern Cape Province of South Africa. In particular, soil erosion modelling that integrates sediment delivery ratios in GIS has not been undertaken. The application of topographic indices is useful for consistent, accurate, low cost (Kheir *et al*., 2007) and broad scale prediction of areas vulnerable to gully erosion. The hypothesis that topographically similar areas have similar hydrologic functions that can be used as surrogates for identifying topographic zones susceptible to gully development is thereby tested. De Santis *et al.* (2010) highlight the importance of soil chemical properties and clay mineralogy in piping and gully erosion. These properties include dispersion, swelling, cracking potential and erodibility. In a review of soil erosion studies in South Africa, Laker (2004) calls for the inclusion of the role soil chemistry in soil erosion studies. Furthermore, while the use of remote sensing has gained attention in mapping soil erosion (Flügel *et al*., 1999; Märker *et al*., 2001; Flügel *et al*., 2003; Taruvinga, 2009), the application of object oriented classification techniques to map soil erosion phenomena has not yet been explored.

Thus the objectives of this study are:

(1) To determine the spatial patterns of soil loss in the Keiskamma catchment using a GIS based RUSLE model that integrates sediment delivery ratios to assess the environmental health status of the catchment.

(2) To identify topographic thresholds and zones susceptible to gully erosion.

(3) To characterize the soil physical and chemical properties and link them to the development of pipes and gullies. In addition, the study also seeks to establish whether significant differences exist in the sodic levels between the A and B soil horizons since the abandonment of cultivation in the 1950s and 60s in the catchment could have elevated sodium levels in the A horizon.

(4) To map gully erosion surfaces and valley infill in ephemeral stream channels using object oriented classification as a means of demonstrating the major sediment transfer processes operating in the Keiskamma catchment. Sediments are transferred mostly from rills and gullies (sediment sources) into ephemeral stream channels which act as sediment sinks.

5.2 Methods

This study applied the RUSLE model in the GIS-based Sediment Assessment Tool for Effective Erosion Control (SATEEC) to estimate soil loss and sediment yield for any location within the Keiskamma catchment using RUSLE input data and the spatially distributed sediment delivery ratio. SATEEC is an ArcView extension developed by Lim *et al*. (2005), which is an effective tool to estimate soil loss and sediment yield.

The RUSLE equation (metric) is defined as:

$$A = R \times K \times L \times S \times C \times P \qquad (1)$$

Where A = annual soil loss (ton $ha^{-1}year^{-1}$), R= rainfall erosivity factor ($MJmmha^{-1}year^{-1}$), K = soil erodability factor (ton h $MJ^{-1}mm^{-1}$), L =slope length factor (-), S= slope steepness factor (-), C = cover-management factor (-), and P = supporting practices (-) (Renard *et al*., 1997).

5.2.1 Parameters used for soil erosion assessment

The derivations of the RUSLE parameters required as inputs in the SATEEC GIS System to predict the average annual rate of soil loss are discussed in this section. The key factors that are explored are: Rainfall-runoff erosivity factor (R), Soil Erodibility factor (K), Slope-length and slope steepness (LS) factors and Cover management factor (C). All datasets were projected to UTM WGS 84 projection system and resampled to a grid resolution of 20m.

Rainfall-Runoff Erosivity Factor (R):

Rainfall is a driver of soil erosion processes and its effect is accounted for by the Rainfall-Runoff Erosivity factor (R) in the RUSLE equation. The R-factor accounts for the effect of raindrop impact and also shows the amount and rate of runoff associated with precipitation events. The R factor is computed as total storm energy (E) time the maximum 30-minute intensity (I30), or EI, and is expressed as the rainfall erosion index (Renard *et al*., 1997). Lack of continuous pluviograph data relating to rainfall intensity motivated the application of the equation established by Wischmeier and Smith (1978) to derive the R factor. Climate data for the Keiskamma catchment were obtained from the Water Research Commission (WRC, 1995a). Rainfall related data for the catchment spanned for a period of 52 years. Rainfall data were imported into ArcView since all the weather stations had co-ordinates. Annual and monthly rainfall data for the Keiskamma catchment obtained over 52 years were used to calculate the R-factor in this study. The equation below developed by Wischmeier and Smith (1978) was used in the computation.

$$R = \sum_{i=1}^{12} 1.735 \times 10^{\left(1.5 \log \frac{p_i^2}{p} - 0.8188\right)} \qquad (2)$$

Where: p_i is the monthly amounts of precipitation and p is annual precipitation. The annual summation of p_i^2/p is called the Fournier equation. In recent years a number of interpolation methods have been developed in GIS that are suitable to model rainfall erosivity. Interpolation methods available in most GIS software include the Inverse Distance Weighting (IDW), Kriging, Spline Polynomial Trend, and Natural Neighbour methods. In this study, the rainfall erosivity values for the different stations were used to interpolate a rainfall erosivity surface using the IDW technique available in ArcGIS 9.0. The IDW interpolation method was selected because rainfall erosivity sample points are weighted during interpolation such that the influence of rainfall erosivity is most significant at the measured point and decreases as distance increases away from the point.

Soil Erodibility Factor (K)

Soil erodibility factor (K) in the RUSLE equation is an empirical measure which expresses the inherent susceptibility of a soil to water erosion as determined by intrinsic soil properties. The K factor is rated on a scale from 0 to 1, with 0 indicating soils with the least susceptibility to erosion and whilst 1 indicates soils which are highly susceptible to soil erosion by water. The factor is defined as the rate of soil loss per rainfall erosion index unit as measured on a standard plot.

A digital soil classification coverage captured from a soil map by the Water Research Commission was supplied by the Department of Water Affairs and Forestry for integration into the USLE computation (WRC, 1995b). Fieldwork was conducted to collect soil samples to determine the particle size distribution of the Mispah, Hutton and Glenrosa soil forms dominant in the Keiskamma catchment. Three random samples were collected for each soil type; a soil map was used to determine the spatial distribution of the soil forms in the field. The co-ordinates for the soil sampling locations were collected using a Global Position System (GPS). Soil erodibility was calculated using the equation (3) developed by Wischmeier and Smith (1978). The equation effectively describes soil erodibility as a function of the complex interaction between sand, silt, and clay fractions in the soil and other factors such as organic matter, soil structure and profile permeability class. In general, soils become less erodible with decrease in silt content, regardless of corresponding increases in the sand or clay fraction (Wischmeier and Smith, 1978).

$$K = [(2.1 \times 10^{-4} (12 - OM) M^{1.14} + 3.25(S - 2) + 2.5(P - 3))/7.59 \times 100] \quad (3)$$

Where

K = soil erodibility factor (tonne.h.MJ^{-1}.mm^{-1}).

OM is soil organic matter content,

M is product of the primary particle size fractions

M = (%silt + %very fine sand) × (100 - %clay)

S is soil structure code

P is permeability class

The average soil erodibility for each soil type was computed and added to the soil classification shapefile database in ArcView 3.3 software. The shapefile was subsequently converted to a 20m grid of soil erodibility. Mispah, Hutton and Glenrosa soil forms were assigned a K value of 0.070574, 0.080306 and 0.0780896 respectively. A soil erodibility map was then developed.

Slope-length (L) and slope steepness (S)

The effect of topography on erosion is expressed by the L and S factors in the RUSLE model. The L and S factors can be computed in GIS using a number of empirical formulae (Wischmeier and Smith, 1978; McCool *et al*., 1987, 1989; Desmet and Govers, 1996; Renard *et al*., 1997). A Digital Elevation Model (DEM) was used to derive the L and S parameters using a Slope Length function available in ArcView SATEEC GIS-software. The L factor expresses the ratio of rill erosion (caused by flow) to interrill erosion (raindrop impact) to find the loss of soil in relation to the standard plot length of 72.6 ft. Renard *et al*. (1997) define slope length as the horizontal distance traversed from the origin of overland flow to the point where deposition occurs (a flattened slope) or runoff concentrates into a defined channel. The slope steepness factor (S) relates to the effect of the slope gradient on erosion in comparison to the standard plot steepness of 9%. The effect of slope steepness is greater on soil loss compared to slope length. This study uses a method proposed by Desmet and Govers (1996) to calculate the L and S factors. Besides interrill and rill erosion, Desmet and Govers (1997) note through field observations that the two dimensional approach of the RUSLE considers ephemeral gully erosion as a product of flow convergence. In this procedure the RUSLE is adapted to a two-dimensional landscape in which the upslope length is substituted by the unit contributing area which is defined as the upslope drainage area per unit of contour length. A 20m Digital Elevation Model created using contours was used to derive topographic variables such as slope length and steepness. The equations developed by Desmet and Govers (1996) were used to calculate L and S in this study are shown below.

L factor:

$$L = \left(\frac{\lambda}{22.13}\right)^{m} \qquad m = \frac{F}{(1+F)} \quad F = \frac{\sin\beta / 0.0896}{3(\sin\beta)^{0.8} + 0.56} \tag{4}$$

$$m = ((\sin([slop] * 0.01745) / 0.0896) / (3 * pow(\sin([slop] * 0.01745), 0.8) + 0.56))$$

Where λ is the slope length along the horizontal projection rather along the sloping surface, m is the slope length exponent and β is slope angle (%). The L factor with upslope drainage contributing area (Desmet and Govers, 1996) was computed as:

$$L(i,j) = \frac{(A(i,j) + D^{2})^{m+1} - A(i,j)^{m+1}}{x^{m} . D^{m+2} . (22.13)^{m}} \tag{5}$$

$$L = \frac{(pow([Flowacc] + 1000, ([m]+1)) - pow(([Flowacc], [m]+1))}{(pow(100, [m]+2) * pow(22.13, [m]))}$$

Where A (i, j) [m] is unit contributing area at the inlet of grid cell, D is grid spacing and x is shape correction factor

The S factor was computed thus:

$$S(i,j) = \begin{cases} 10.8\sin\beta(i,j) + 0.03, \tan\beta(i,j) < 0.09 \\ 16.8\sin\beta(i,j) - 0.50, \tan\beta(i,j) \geq 0.09 \end{cases} \tag{6}$$

Where β (*i, j*) is the mean slope angle of all sub-grids in the steepest direction (McCool *et al*., 1987, 1989). Hillslope length λ is calculated as the grid area divided by the total length of streams in the same grid.

Cover management factor

The effect of vegetation cover as a control on soil erosion is well established. Vegetation is regarded as the second most critical factor after topography (Benkobi *et al*., 1994; Biesemans *et al*., 2000). In the RUSLE model, the effect of vegetation cover is incorporated in cover management, the C factor. The application of the Normalized-Difference Vegetation Index derived from remotely sensed images has been proved to be useful in providing an estimate of the vegetation cover management factor. The NDVI (Near Infrared-Red)/ (Near Infrared+ Red) is a robust vegetation index which has been applied successfully in studies relating to vegetation dynamics. Landsat 5 Thematic Mapper satellite data acquired on 12 December 2006 was used to derive the NDVI by computing the ratio (Band 4 - Band 3)/ (Band 4 + Band 3). The NDVI is highly correlated with the amount of green biomass, and can therefore be applied successfully to provide information relating to the green vegetation variability. Studies by Van der Knijff (1999, 2000) and Van Leeuwen (2003, 2005) provide a more refined and reasonable estimation of the C-factor using the NDVI. The Landsat 5 TM image was accurately orthorectified and terrain corrected using satellite orbital math modelling method which applies the Toutin's Low Resolution Model available in the PCI Geomatica orthoengine software (PCI Geomatica 10.3, 2009). The Universal Transverse Mercator (UTM) projection in WGS84 was used in the co-registration. A 2,5m geocoded panchromatic SPOT band for the area was used as the reference image and a 20 m resolution DEM was used to correct for the topographic distortions. The cubic convolution resampling method was used for orthorectification. Rectification errors were less than 0.35 pixels (RMSE). Atmospheric corrections using the Quick Atmospheric Correction algorithm available in ENVI were applied to Landsat 5 TM image to improve the spectral fidelity of the satellite data. Accurate orthorectification of digital satellite imagery ensured that Landsat 5 TM and other ancillary datasets overlaid perfectly. The following equation was used to derive the C-factor in this study.

$$C = \exp\left[-\alpha.\frac{NDVI}{(\beta - NDVI)}\right] \quad (7)$$

Where α, β parameters determine the shape of the NDVI curve. Reasonable results are produced using values of $\alpha = 2$ and $\beta = 1$.

The hypothetical relationship between the NDVI and C-factor according to the exponential scaling formula is illustrated by Figure 5.1.

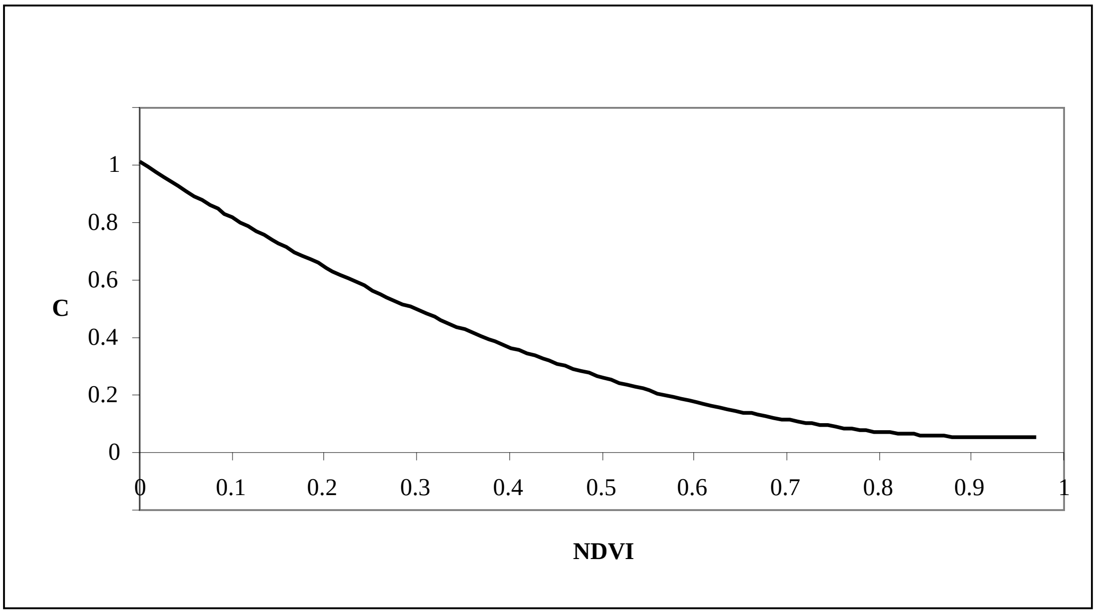


Figure 5.1 Relationship between NDVI and RUSLE-C (Van Leeuwen , 2005)

Vegetated areas usual have NDVI values much greater than 0.1 while values less than 0 rarely contain vegetation and relate to non-photosynthetic materials such as water and bare soil. A lower vegetation threshold of 0.05 was set, below which vegetation was envisaged to be absent.

Conservation practice P factor

The conservation practice P factor is an important consideration of the RUSLE model. The support practice factor is defined as the ratio between soil loss with a specific support practice and the corresponding loss with upslope and downslope tillage. Renard and Forster (1983) explain that support practice essentially affects soil erosion through altering the flow pattern, gradients, or direction of surface runoff and by reducing the amount and rate of runoff. Information regarding conservation was obtained through field observations in the Keiskamma using a GPS. Shape files for protected areas which receive high priority in terms of conservation practice were downloaded from the South African Biodiversity Institute website (SANBI, 2009). These conservancy zones were assigned a P factor of 0.001, reflecting stringent conservation practice in these areas. Field assessments in the rest of the Keiskamma catchment revealed that no significant conservation practices were in place and a P factor of 1 was assigned to them. The conservation rating ranges from 0.001 to 1, with a lower P-value indicating that a more effective conservation practice is in place to curtail soil erosion. Overgrazing, thicket degradation, and general neglect of the environment in the communal areas are evident in most parts of the catchment. Bennett and Barrett (2007) studied the grazing management systems in some communal areas in the Central Eastern Cape including parts of the Keiskamma catchment and identified three grazing management systems. The scholars identified the open access system, where there is complete lack of grazing management, the controlled system, where grazing is governed by the community and lastly private grazing, whereby the landowner takes responsibility for the grazing on their private land. Field observations in the catchment indicated that open access and a loose form of community controlled system seem to be operational in the Keiskamma catchment.

Sediment delivery ratios

An area based method developed by Vanoni (1975) was used to estimate the SDR in the SATEEC GIS system. This method uses a generalized curve derived from experimental work in 300 watersheds. The watershed area at any point in the catchment is computed from the flow accumulation map, which is derived from the DEM pre-processing to compute the LS factor (Lim *et al*., 2005). The power function used to develop the generalized SDR curve is shown below.

$$SDR = 0.4724A^{-0.125} \tag{8}$$

where, A= watershed area (km^2).

5.2.2 Accuracy assessment of soil loss

Model validation was done to ascertain the quality of results produced by the SATEEC model and test the usefulness of the model to predict soil loss. The Kappa Analysis Tool extension developed by Jenness and Wynnes (2005) in ArcView 3.3 was used for accuracy assessment in this study. The application is based on the accuracy assessment theory presented by Congalton and Green (1999). The Kappa Analysis method, widely used in remote sensing accuracy assessments is a powerful method used to measure the agreement between predicted and observed phenomena (Jenness and Wynnes, 2005). This study applies Cohen’s Kappa to assess the accuracy of the SATEEC soil loss results. Extensive fieldwork was carried in the Keiskamma catchment to randomly identify eroded and non eroded sites and their locations were captured using a GPS. The SATEEC soil loss classes were reclassified into two; low soil loss and high soil loss. The very low and low soil loss classes were reclassified into low soil loss and the moderate to extremely high soil classes were reclassified into the high soil loss classes (after Le Roux *et al*., 2008). The reclassified soil loss results were then compared with field validated sample points in ArcView 3.3 software.

5.2.3 Mahalanobis distance method

Mahalanobis distance method has been applied in this study to compute the topographic thresholds and determining the susceptibility clusters for gully erosion. This method is based upon the mean and variance of the predictor variables as well as the covariance matrix of all the variables, consequently utilizing the covariance among variables (Jenness, 2003). Mahalanobis distance is the resultant of the quadratic multiplication of mean difference and inverse of the joint covariance matrix. A total 559 gully locations were collected for the Mahalanobis distance analysis. Of these, 50 were located using a GPS in the field while the rest were collected from 2.5m pansharped SPOT 5 satellite imagery and aerial photography. The analysis was executed in Arc View 3.3 software with the Mahalanobis extension. The mathematical expression to compute Mahalanobis distances is:

$$D^2 = (\chi - \eta)^T C^{-1} (\chi - \eta) \quad (9)$$

Where:

D^2 = Mahalanobis distance

χ = Vector of data

η = Vector of mean values of independent variables

T = Inverse Covariance matrix of independent variables

C^{-1} = Indicates vector should be transposed

A 20m DEM was used to derive slope, topographic wetness index, stream power index, and planiform curvature for application in the Mahalanobis computation. Topographic wetness index (*TWI*) and stream power index (*SPI*) are derived using the following formula:

$$TWI = In(A_s / \tan\beta) \quad (10)$$

$$SPI = In(A_s * \tan\beta) \quad (11)$$

Where

A_S = Specific catchment area (local upslope contributing area) (m^2)

β = local slope.

Mahalanobis distances do not have upper limits and were rescaled to a range between 0 and 1 by conversion to Chi-square p-values to facilitate analysis. The Mahalanobis distances were converted to p-values using 3 degrees of freedom. A p-value close to 0 indicates high Mahalanobis distance value, suggesting conditions divergent to the ideal combination of predictor variables whilst p-values close to 1 indicate low Mahalanobis distances reflective of conditions matching to the ideal combination of predictor variables. The probability map was further reclassed in 5 gully susceptibility classes. Gully susceptibility was rated into very low, low, moderate, high and very high.

5.2.4 Physical and chemical characterization of soils in badlands

Field observations were done to assess the physical conditions of the soils in areas experiencing extensive gully erosion. Soil samples were collected at 24 gully sites which showed evidence of severe piping. SPOT-5 panchromatic satellite imagery was used in the preliminary selection of gullied sites. Soil sampling was done in the A and B horizons along the slope profile on three topographic positions; top, middle and bottom. The A horizon was sampled up to a depth of 30cm and the B horizon was sampled from 30cm to 60cm.

The samples were taken to the Döhne Analytical Services soil laboratories in Stutterheim, Eastern Cape, for chemical and textural analysis. The chemical analysis was done to determine the geochemical characteristics which affect soil dispersion such as the sodium adsorption ratios, sodium exchangeable percentage, salinity, cation exchange capacity, concentration of bases, and the base saturation. The analytical techniques applied in this study were based on standard methods used in Southern Africa as described by Watson *et al.* (1984) and Walker (1997). Soil organic carbon was also analysed using the Walkey-Black method. The USDA texture ternary diagram was plotted using the DPlot software. Soil erodibility was calculated using equation 3 described earlier. Electrical conductivity as an indicator of salinity was measured in mS/m. The concentration of bases was measured in milli- equivalent elements per litre of the soil/paste extract and later converted to $cmol_c\ kg^{-1}$. CEC is expressed as centimoles of positive charge per kilogram of soil cmol (+)/kg or $cmol_c$/kg. Christidis (1998) indicates swelling capacity of soils increases with CEC. Other parameters which were analysed include the SAR, ESP, percentage base saturation and CEC clay. The formulae used to derive the parameters are shown below:

$$SAR = \frac{[Na^{+}]}{\sqrt{(0.5[Ca^{2+}] + 0.5[Mg^{2+}])}} \quad (12)$$

Where $[Na^{+}]$, $[Ca^{2+}]$, and $[Mg^{2+}]$ the concentrations (in mmol of charge per litre) of the sodium, calcium, and magnesium ions on the soil solution.

$$ESP = \frac{Exchangeable\ sodium, cmol_c / kg}{Cation\ exchangeable\ capacity, cmol_c / kg} \times 100 \quad (13)$$

$$\%\ base\ saturation = \frac{\sum cmol_c\ of\ \ primary\ exchangeable\ bases}{cmol_c\ of\ CEC} \times 100 \quad (14)$$

$$CEC\ Clay = CEC\ Soil \times \frac{100}{\%\ Clay} \quad (15)$$

5.2.5 Mapping Valley Infill and Erosion features

The valley infill phenomenon is widespread in the degraded ephemeral streams of the central Keiskamma catchment where sediment accumulation is manifest (see Figure 5.10). Object oriented classification was used to map valley infill within ephemeral stream channels and erosion features such as gullies. Sediment accumulation in ephemeral stream channels (class 1) can be reliably detected using remote sensing. Other land cover types that were classified include roads (class 2), erosional surfaces (class 3), mixed forest (class 4), sparse and degraded vegetation (class 5). A pan sharpening algorithm called Principal Component Spectral Sharpening was used for the fusion of Landsat 5 TM and SPOT 5 panchromatic band to enhance the spatial resolution of Landsat 5 TM.

The hybrid fused image possesses both the high spectral resolution of Landsat 5 TM and a high spatial resolution of 2.5m inherited from the SPOT 5 panchromatic band. Object oriented image classification was achieved by first applying multiresolution segmentation at a scale parameter of 20 before applying the hierarchical classification in Definiens Developer software (Definiens 2009). Layer brightness was used to separate bare areas such as eroded surfaces and roads. The length/width ratio was then used to separate roads from erosion features, as the former are more elongated than the latter. Valley Infill and dense vegetation were classified using the NDVI; vegetation vigour within ephemeral channel valley infill is higher than on hillslopes and adjacent areas. The separation of mixed forest from sparse and degraded vegetation was also done using the NDVI thresholding of image objects.

The brightness parameter is calculated as follows:

$$\bar{c}(v) = \frac{1}{w^B}\sum_{k=1}^{K} w_K^B \bar{c}_K(v) \qquad (16)$$ (Definiens 2009).

Where

w_K^B = brightness weight of image layer k with

$$w_K^B = \begin{cases} 0 \\ 1 \end{cases}$$

K = number of image layers k for calculation

w^B = sum of brightness weight of all image layers k used for calculation with

$$W^B = \sum_{k=1}^{K} W_K^B$$

$\bar{c}_K(v)$ = mean intensity of image layer k of image object v.

$\bar{c}_k^{\min}$ = darkest possible intensity value of image layer k.

$\bar{c}_k^{\max}$ =brightest possible intensity value of image layer k.

Accuracy assessment was done to validate the classification using ground reference data collected using a GPS.

5.3 Results and interpretation

5.3.1 Soil Loss

Results for the RUSLE factors which were computed in this study are presented in Figures 5.2 up to 5.6. The SATEEC RUSLE approach effectively illustrates the spatial distribution of soil loss throughout the Keiskamma catchment. The soil loss distributions in the catchment are illustrated by Figure 5.7 and the proportions are summarized in Figure 5.8. The total soil loss in the Keiskamma catchment is 9.27 x 10^6 ton/year over an area of 257121 hectares. The mean soil loss in the Keiskamma catchment is 36.063 tons/hectare/year. A study by Le Roux *et al*. (2008) indicates that the average predicted soils loss for South Africa is 12.6 tons/hectare/year. Their study further reveals that the Eastern Cape Province has the highest annual soil loss contribution of 28% with a soil loss rate of 25 tons/hectare/year. Soil loss tolerances proposed for South Africa range from 3 tons/hectare/year for shallow soils and 10 tons/hectare/year for deep alluvial soils (McPhee and Smithen, 1984). The results indicate that up to 47% of the catchment has soil losses higher than 12 tonnes/ha/yr. It is evident that the rate of soil loss in the Keiskamma catchment is way above sustainable tolerance limits. The remaining proportion of the catchment experiences very low to low soil losses, largely due to the role of vegetation in the form of forest plantations and other conservancy areas.

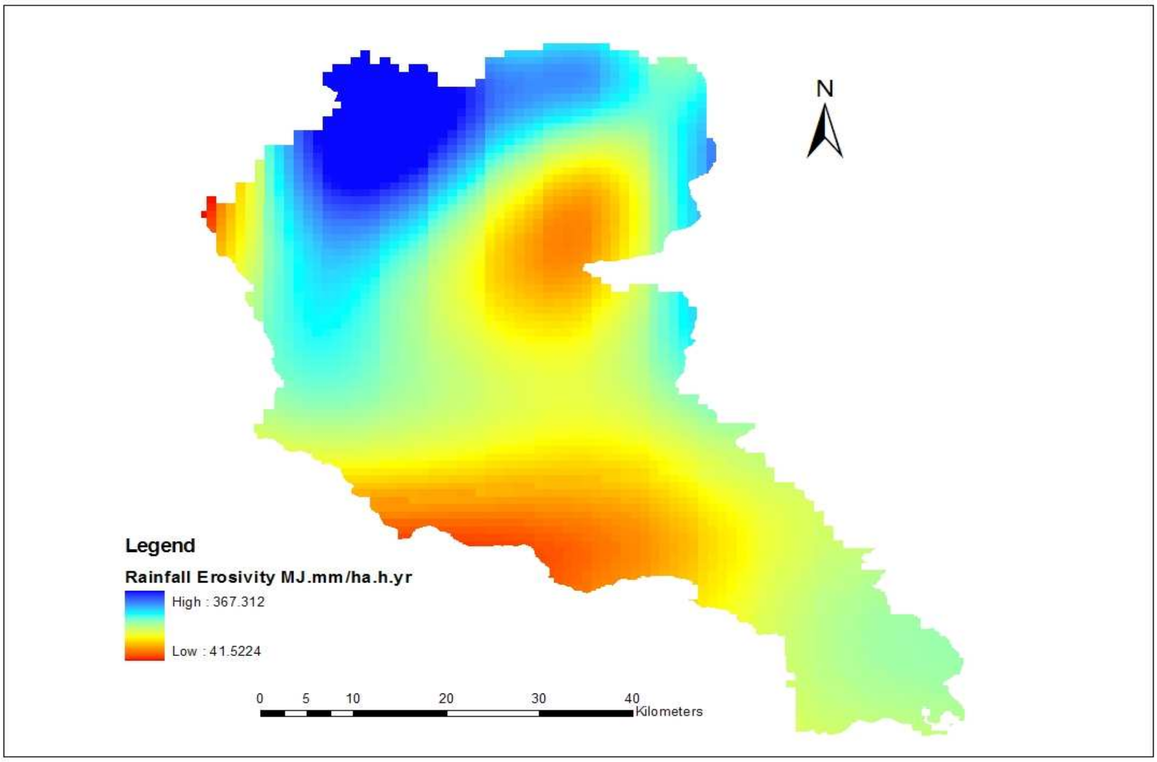


Figure 5.2 Rainfall erosivity factor

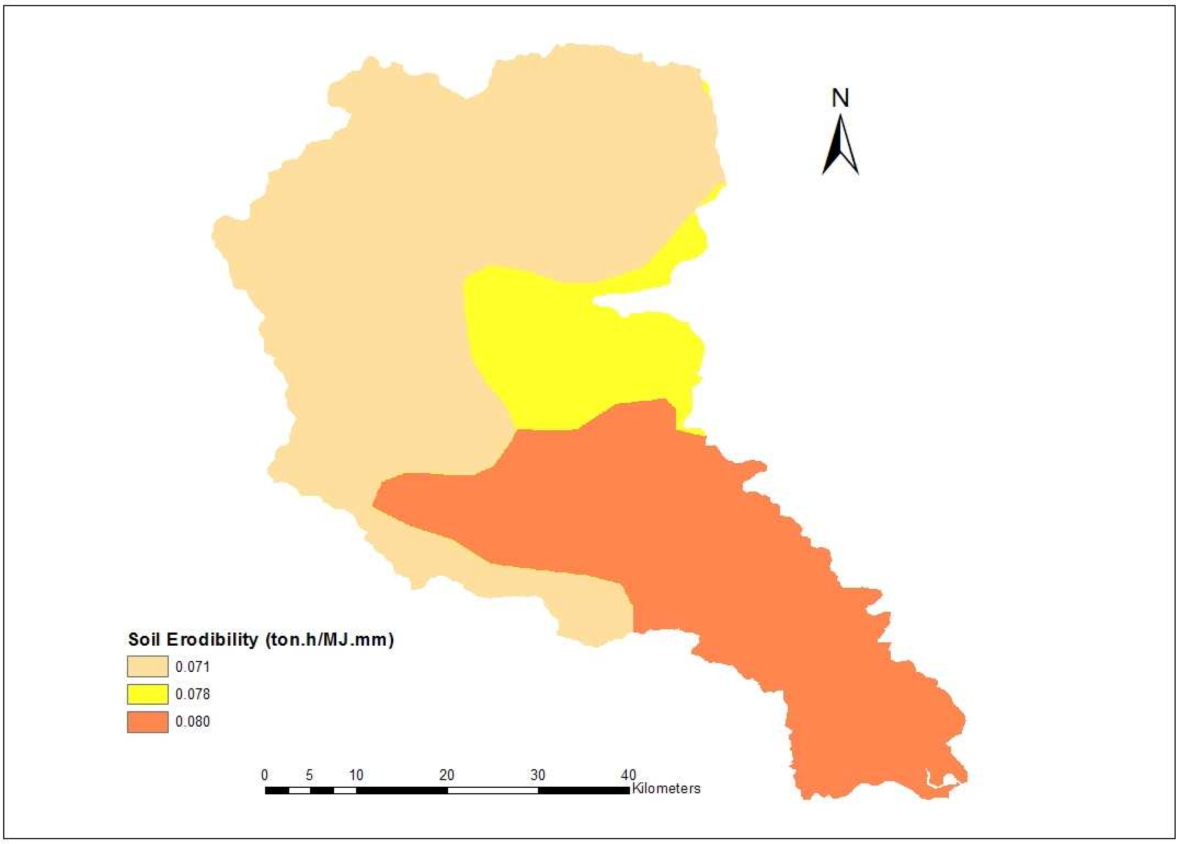


Figure 5.3 Soil erodibility factor

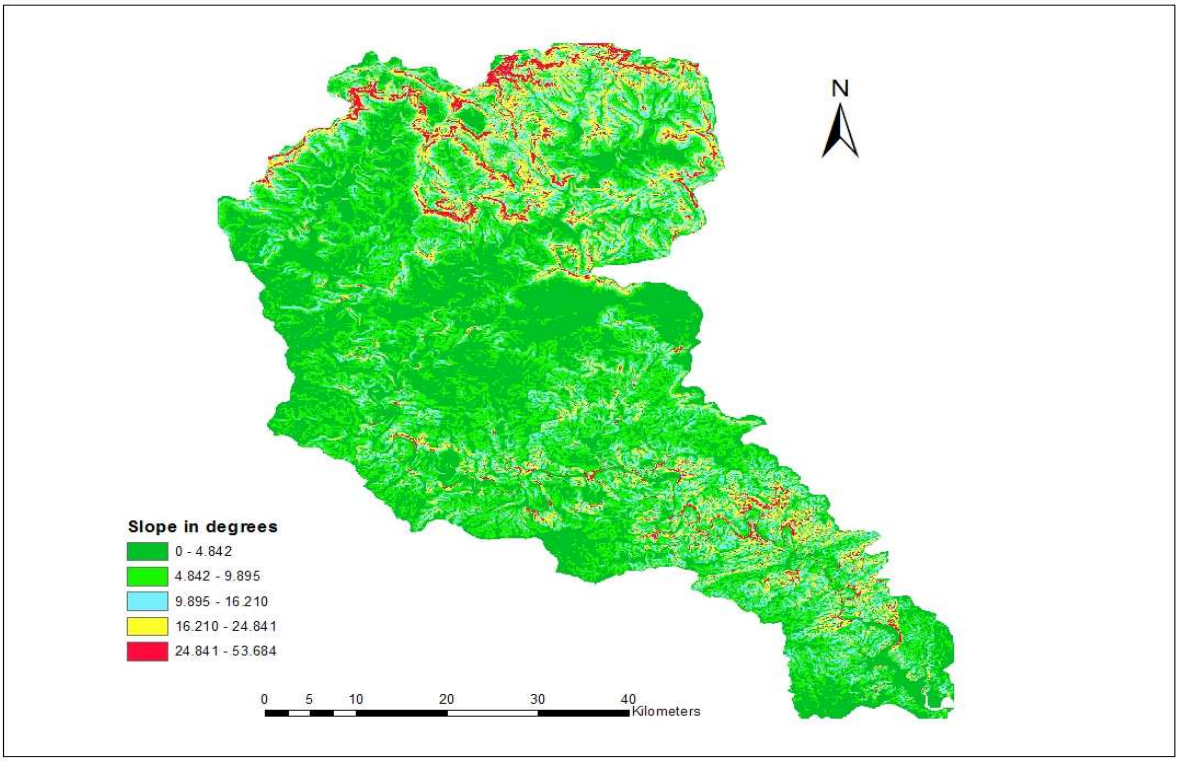


Figure 5.4 Slope

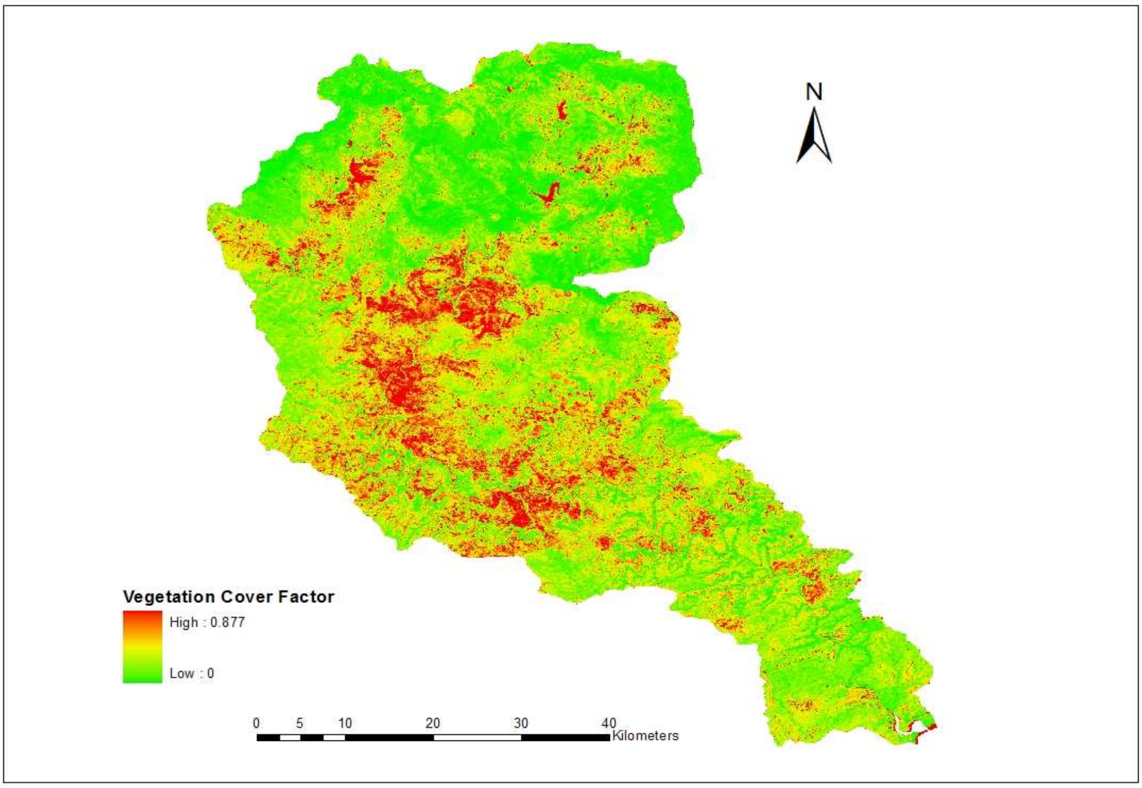


Figure 5.5 Vegetation cover factor

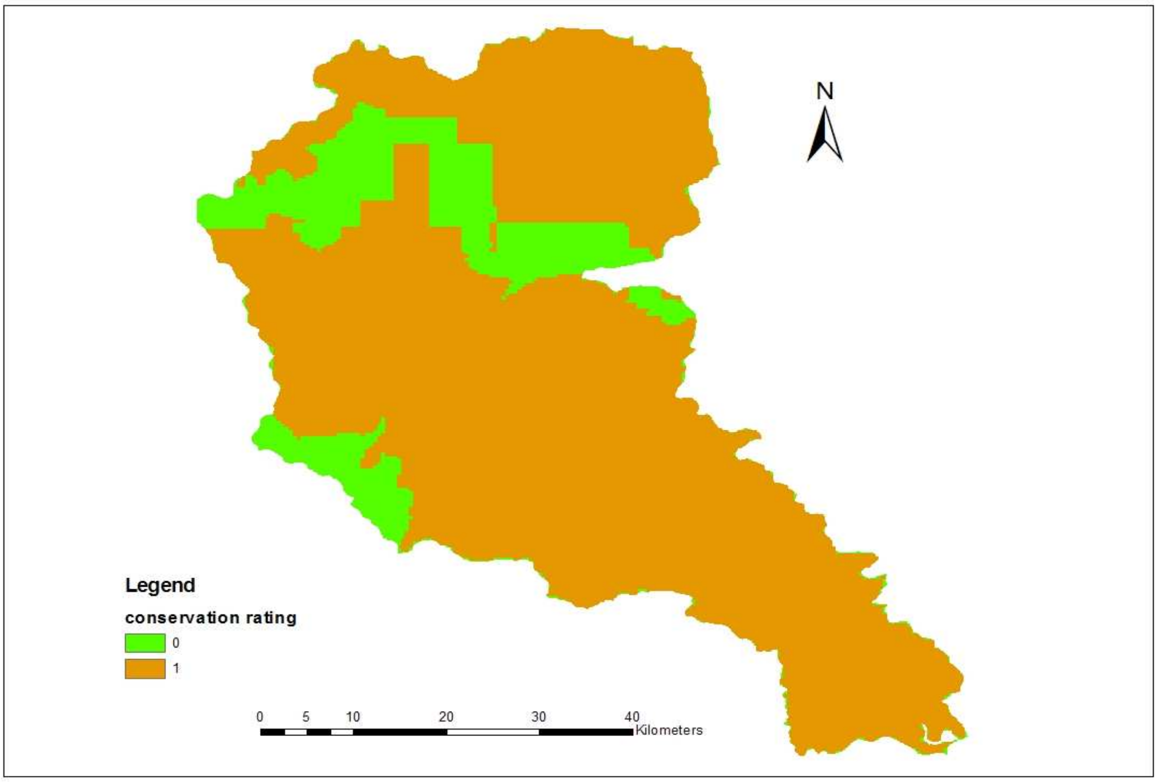


Figure 5.6 Conservation practice factor

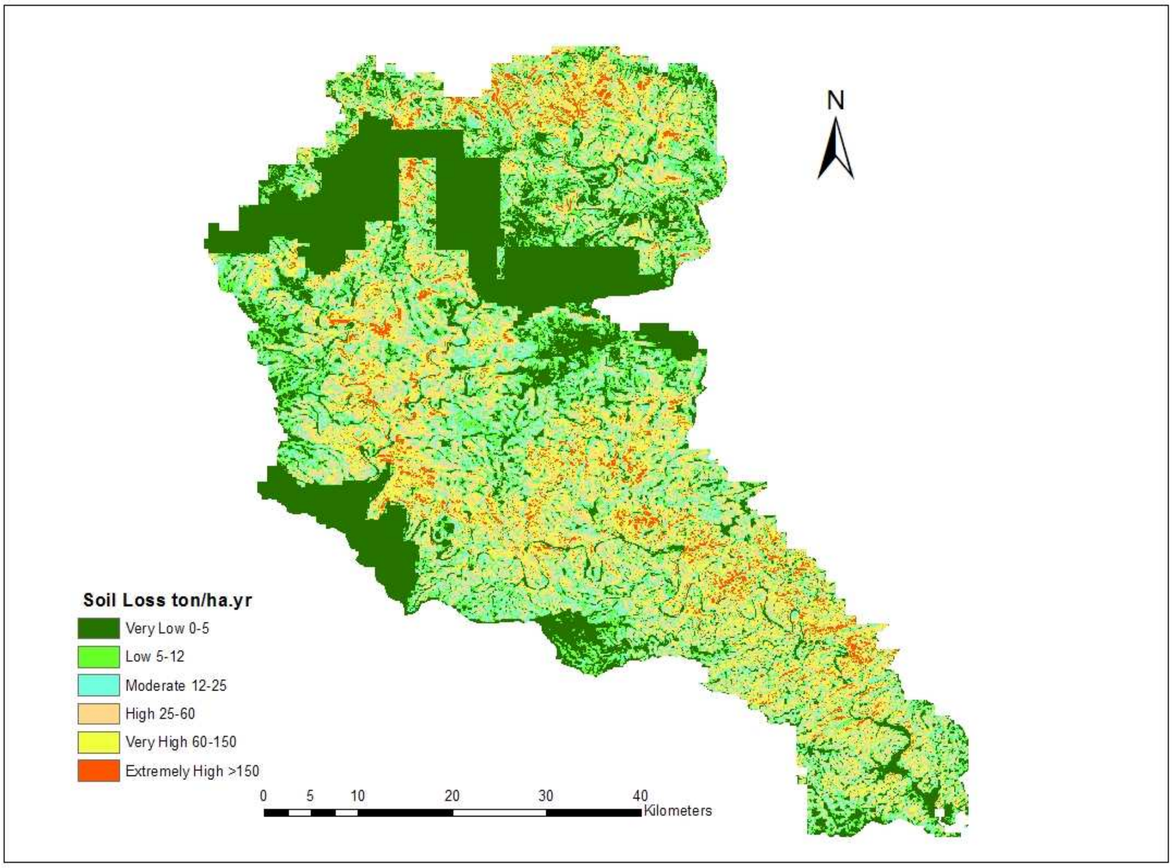


Figure 5.7 Soil loss distribution in the Keiskamma catchment

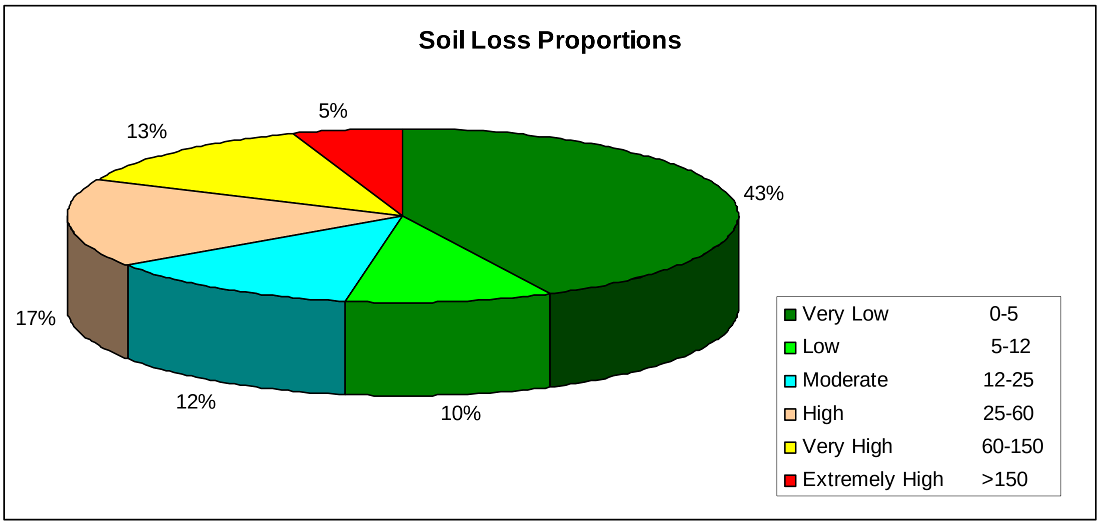


Figure 5.8 Soil loss proportions in the Keiskamma catchment

5.3.2 Model validation

The accuracy assessment results for the soil erosion risk assessment are shown in Tables 5.1 to 5.4. The proportion error matrix (Table 5.1) shows the proportional classification successes along the diagonal and the proportional misclassifications in the upper and lower triangles. The accuracy report in Table 5.2 summarizes the producer's accuracy, user's accuracy, sensitivity and the specificity of each class (Jenness and Wynnes, 2005). The error report is shown in Table 5.3; it summarizes the omission and commission error. Table 5.4 shows the overall accuracy, Kappa Statistics; the KHAT is the chance-corrected measure of model accuracy, calculated on the actual agreement between predicted and observed values and the chance agreement between the row and column totals for each classification (Jenness and Wynnes, 2005). The Z-score of 4.129 and associated *P*-value of 0.0000182 (Table 5.5) reveal the probability that the SATEEC model performs better than the random chance at predicting the occurrence of erosion on the landscape. It can therefore be concluded that results of the SATEEC model are a good indication of the soil loss distribution in the Keiskamma catchment.

Table 5.1 Proportion error matrix

Class	**No Erosion**	**Erosion**	**SUM**
No Erosion	0.256	0.070	0.326
Erosion	0.140	0.535	0.674
SUM	0.395	0.605	1.000

Table 5.2 Accuracy report

ID	PRODUCER	USER	SPECIFICITY	PRED. POWER	ID CLASS
1	0.647	0.786	0.885	0.793	No Erosion
2	0.885	0.793	0.647	0.786	Erosion

Table 5.3 Error report

ID	OMISSION ERROR	COMMISSION ERROR	ID CLASS
1	0.353	0.115	No Erosion
2	0.115	0.353	Erosion
Overall	0.209		

Table 5.4 Summary of overall statistics

SUMMARY OF OVERALL STATISTICS	
Overall Accuracy/Sensivity:	0.791
Overall Misclassification:	0.209
Khat:	0.548
Variance:	0.018
Z:	4.129
P:	0.0000182

5.3.3 Topographic thresholds

The results of Mahalanobis Distances computation are presented in this section. The following variance/covariance matrix and inverse covariance matrix were computed in this study.

Variance/Covariance Matrix   Inverse Covariance Matrix

$$
\begin{array}{cccc}
\text{TWI} & \text{Slope} & \text{SPI} & \text{Curvature}
\end{array}
\qquad
\begin{array}{cccc}
\text{TWI} & \text{Slope} & \text{SPI} & \text{Curvature}
\end{array}
$$

$$
\begin{pmatrix}
4.1335 & -4.0191 & 1.7825 & -0.1940 \\
-4.0191 & 18.8709 & 0.2489 & -0.0830 \\
1.7825 & 0.2489 & 1.1958 & -0.1307 \\
-0.1940 & -0.0830 & -0.1307 & 0.0481
\end{pmatrix}
\begin{pmatrix}
2.2330 & 0.5233 & -3.3490 & 0.80938 \\
0.5233 & 0.1760 & -0.7863 & 0.2780 \\
-3.3490 & -0.7863 & 6.2131 & 2.0211 \\
0.8094 & 0.2780 & 2.0211 & 30.0555
\end{pmatrix}
$$

Topographic thresholds susceptible to gully erosion are shown in Table 5.5. The results indicate that gully erosion occurs predominantly in low slope angles with a mean vector of 6.838°.

Table 5.5 Topographic thresholds

Topographic Parameter	*Mean Value Vector*
TWI	4.869
Slope	6.838
SPI	1.063
Planform Curvature	-0.070

This value is below the average slope for the Keiskamma catchment which is 8.497°. This indicates that lower slopes are more vulnerable to gullying and that most gullies occur in areas of high topographic wetness with a mean vector of 4.869. This is higher than the average topographic wetness of 4.041 for the catchment. Gully erosion is also dominant in areas of higher stream power index; the mean vectors for gully occurrence is 1.063 and yet mean stream power index for the catchment is 0.723. The predominance of gullies on concave slopes is notable, given the mean vector of -0.070, which depicts concavity. The topographic thresholds derived in this study are largely consistent with the results reported by Kakembo *et al*. (2009) in the adjacent communal areas of Ngqushwa. The study confirms the predominance of gullies of in lower concave slopes of 5-9°. Gully heads emerge once a critical soil surface slope is surpassed (Poesen, 2002).

Susceptibility to severe forms of erosion

Figure 5.9 shows the topographic zones susceptible to severe forms of erosion, particularly gullying. Values close to 1 are highly susceptible to gully erosion whilst those close to zero are less likely to be eroded.

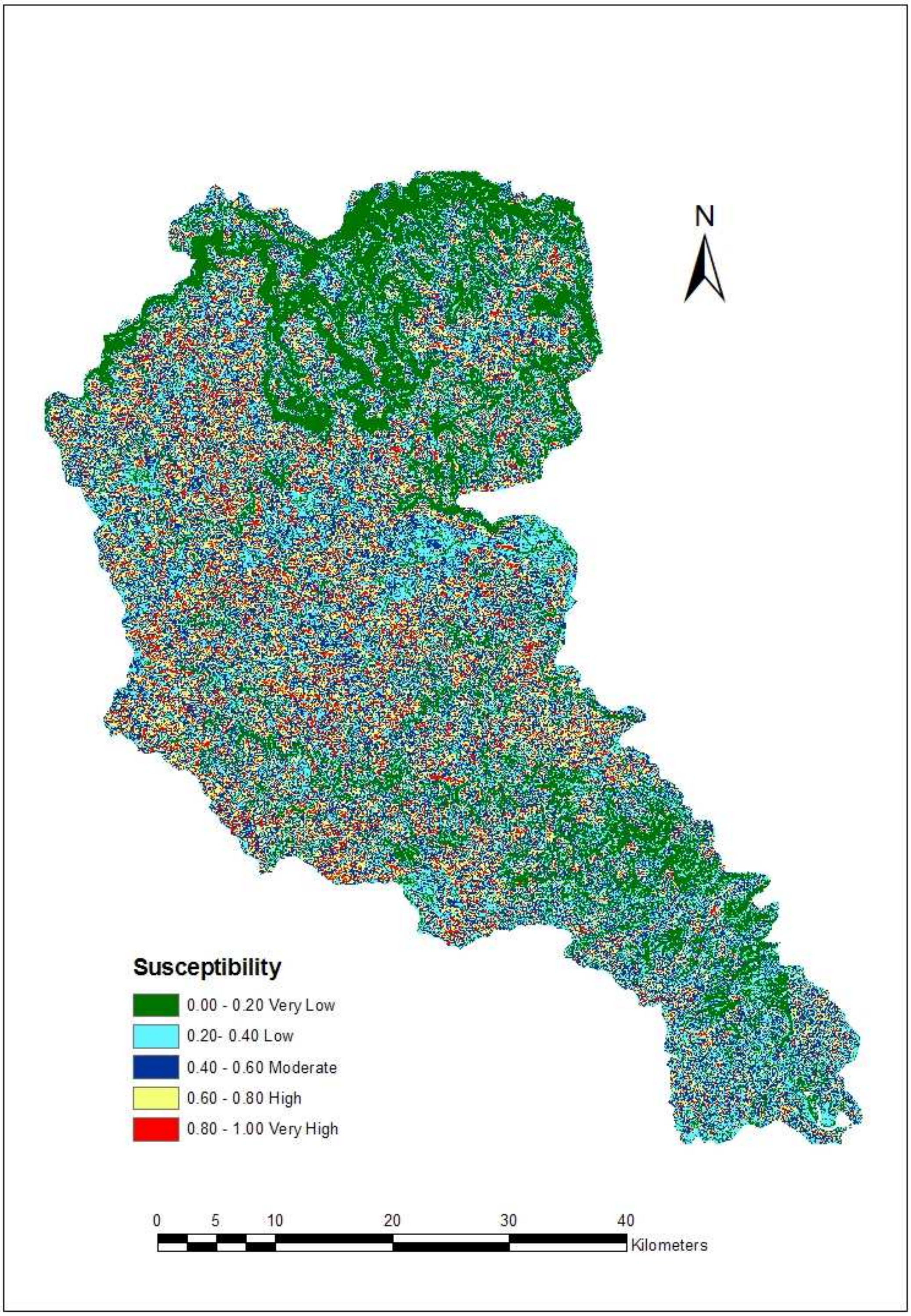


Figure 5.9 Topographic zones susceptible to gully erosion

5.3.4 Soil analysis results and interpretation

Site appraisal

Piping or tunnel erosion was noted as one of the main factors accelerating the development of badlands as shown in Figure 5.10. Soil colour varied from grey, greyish brown to reddish brown.

Figure 5.10 Piping and gully erosion in Keiskamma in June 2009.

Horizon A soils were friable and exhibited a weak fine granular, massive structure and were mostly derived from colluvial mudstone. Crusting and capping was evident in greyish A-horizon soils. Small and irregular blocks of fine-grained grey and red mudstone which break easily were manifest in most badlands. Soils from the B horizon displayed variations of weak sub-angular blocky and massive structure. The exposed B horizon comprised mostly red-brown mudstone with rounded caps and a columnar structure. Calcrete, iron and manganese concretions were a common feature in the badlands.

Physico-chemical properties

Table 5.6 and 5.8 show the soil chemical and physical properties respectively. To characterise the dispersive site morphology of the sampled sites, diagnostic analysis was executed using the functional relationships between physico-chemical parameters such as EC and ESP, % clays and SAR, EC and % Base saturation, pH and SAR (Figure 5.11 up to 5.14). These physico-chemical parameters are referred to as diagnostic site signatures because of their ability to characterize the dispersive state of soil. The relationship between EC and ESP was investigated by Rengasamy *et al*. (1984) and was found useful in predicting soil dispersion. Faulkner *et al*. (2000) also demonstrated that a pH and SAR signature is useful in investigating the extent of material buffering as dipersivity changes at a single site.

The scatterplot (Figure 5.11) for EC and ESP indicates that the soils are mostly sodic and non-saline. The analysis used an ESP threshold of 5% and salinity threshold of 200 mS/m above which soils are regarded as sodic and saline in South Africa and parts of Australia. The scatterplot for EC vs ESP reveals that most of the soil samples in the A and B horizon had ESP values ranging from 5 to 15%, elevated levels of sodicity and salinity are distinct in the B-horizon with ESPs reaching 35% in areas dominated by piping, with salinity levels of 887 mS/m. The SAR vs % Clay scatterplot (Figure 5.12) reveals that most soil samples surpass the SAR threshold of 2 and clay content of 10%. An increase in the clay content above 10% does not however directly impact soil dispersion; the type of clay mineral present and SAR values are more significant. In South Africa, soils become dispersive when they have SAR values exceeding 2 but also need to have sufficient clay of at least 10% to supply colloids which support dispersion. The CEC values suggest the presence of 2:1 silicate clays such as illite and smectites which are highly dispersive. The scatterplot for SAR vs % Clay also shows that the highest SAR values are found within the midslopes, whilst higher clay percentages are predominant in the upper slopes. Notwithstanding a few highly elevated SAR values in the midslopes, SAR values do not differ significantly with slope position.

Table 5.6 Chemical properties of the soil samples

S#	SH	Slope Position	pH	EC_{25}	Ca	K	Mg	Na	CEC	CEC Clay	SB	%BS	%ESP	SAR
1	B	Lower	6.2	45	3.7	1	4.9	1	7.5	26.1	10.6	141.9	13.3	2.89
2	A	Middle	5.9	230	4.5	1.1	2.5	1	11.1	59.2	9.1	81.7	9	3.04
3	B	Upper	5.7	39	3.4	1	5.2	1	11.4	23.9	10.6	93.1	8.8	2.74
4	A	Upper	5.8	20	3.9	1.1	2.8	0.8	11.2	63.6	8.6	77.1	7.1	2.19
5	B	Middle	5.5	128	2.6	1.1	2	0.8	6.6	36.4	6.5	98	12.1	3.57
6	B	Upper	6.8	1191	3.9	1.2	6.5	0.8	17.6	10.4	12.4	70.5	4.5	5.79
7	B	Lower	7.3	51	3	1.1	2.1	0.9	7.2	52.5	7.1	98.8	12.5	2.72
8	A	Upper	5.3	101	2.7	1.1	1.9	0.8	6.7	38.2	6.5	96.7	11.9	3.72
9	B	Middle	5.2	887	3.5	1.1	6	6.6	18.8	8.1	17.2	91.5	35.1	7.2
10	B	Middle	7.2	280	6.2	1.2	5.4	2	19.7	13.0	14.8	74.9	10.2	8.84
11	B	Lower	6.3	60	2.7	1	2.4	0.9	7.3	31.1	7	96.4	12.3	3.62
12	B	Upper	5.1	51	1.3	1	1.5	0.7	5.1	33.0	4.5	88	13.7	2.63
13	A	Middle	7.4	19	4	1.2	1.9	0.7	8.1	144.2	7.8	95.9	8.6	1.46
14	B	Middle	7.5	78	5.2	1.3	3.3	1	9.3	45.4	10.8	115.9	10.8	3.47
15	A	Lower	6	21	2.5	1.3	2	0.6	6.1	83.9	6.4	105.6	9.8	1.49
16	A	Lower	6.4	28	3.9	1.2	3.1	0.8	10.4	50.9	9	86.3	7.7	2.47
17	A	Upper	6	17	2.5	1.4	1.6	0.7	9.7	104. 9	6.2	63.6	7.2	1.49
18	A	Lower	7.8	118	3.8	1.2	2.4	1.1	12.1	36.6	8.5	69.9	9.1	4.32
19	B	Upper	6.3	18	4.6	1	3.6	0.9	19.1	47.3	10.1	52.7	4.7	2.7
20	B	Lower	5.2	282	3.9	1.2	6.4	1.8	5.9	10.4	13.3	224.7	30.5	5.88
21	A	Upper	4.8	59	1.3	1.1	1.6	0.7	4	36.6	4.7	118.3	17.5	2.22
22	A	Middle	5.5	26	1.6	1	1.6	0.7	8	42.6	4.9	60.8	8.8	2.35
23	B	Lower	6.7	13	3.6	1.2	1.9	0.6	7.9	108.3	7.3	92.4	7.6	1.75
24	A	Middle	7.2	45	4	1.1	1.6	0.6	9.7	173.6	7.3	74.9	6.2	1.44

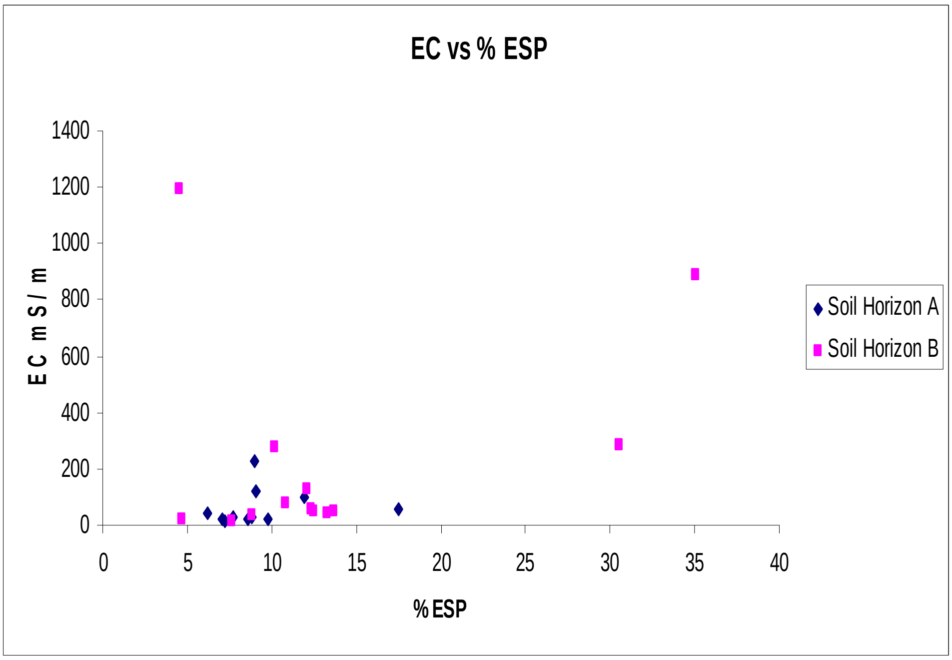


Figure 5.11 Functional relationship: EC vs % ESP

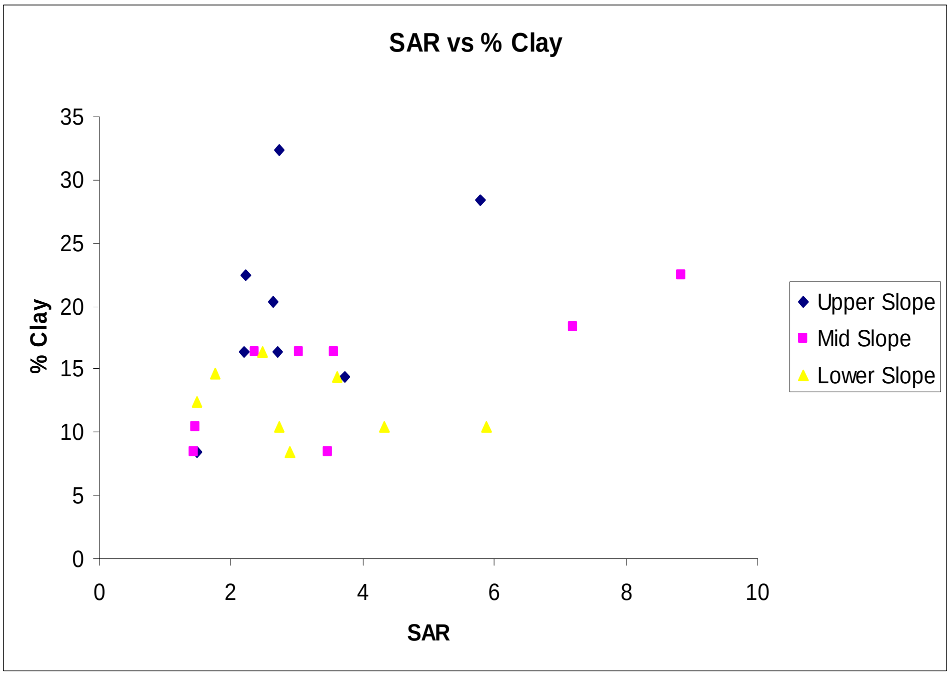


Figure 5.12 Functional relationship: SAR vs % Clay

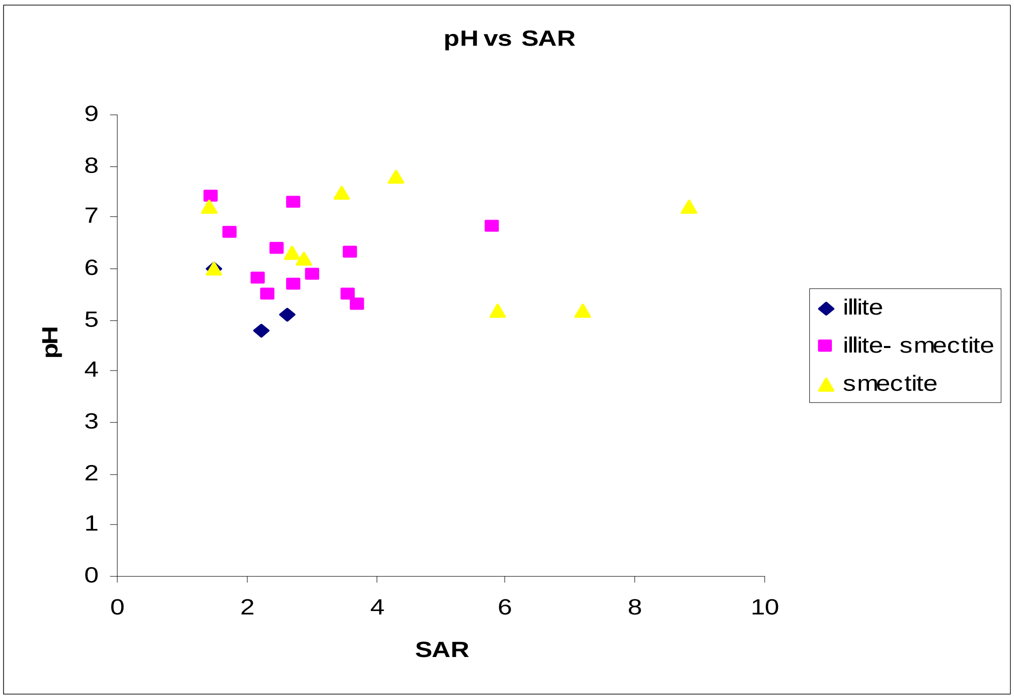


Figure 5.13 Functional relationship: pH vs SAR

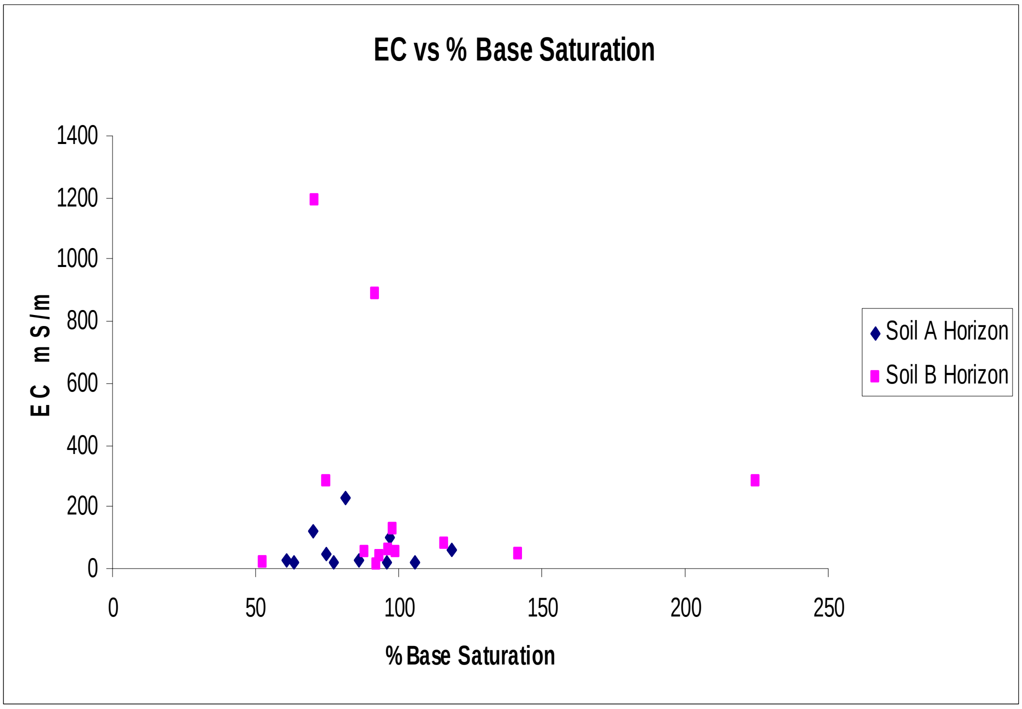


Figure 5.14 Functional relationship: EC vs % Base saturation

Faulker *et al*. (2000) point out that the relationship between pH and SAR shows the buffering role of calcium and its effect to the auto-stabilisation process which results in the reduction of surface SAR, and the flocculation-dispersion behaviour of the clay minerals. The pH vs SAR scatterplot (Figure 5.13) shows that most soil samples surpass the threshold of sodicity (SAR > 2) whilst exhibiting slight acidity (pH less than 7.5). Weak acidity is not strange in dispersive and sodic soils and is probably because of the high dissociation of H^+ ions from smectite clays. The pH results are in accordance with an observation made by Ingles and Aitchison (1969) that weakly acid soils are susceptible to dispersion. The scholars further allude to tunnelling and subsidence which have been observed in weakly acid soils of pH 5. Brady and Weil (2008) point out that smectite dominated soils exhibit lower pH values compared to Fe and Al oxides at the same percent acid saturation.

The graphical relationship between salinity and percent base saturation in Figure 5.14 shows that most of the soils sampled in the Keiskamma have low soluble salt concentrations (EC < 200 mS/m) and variable percentage base saturation. The percentage of base saturation gives an indication of the amount of leaching in the soils, which reduces percentage base saturation, whilst maintaining a constant CEC. The results of the scatterplot of EC vs % Base Saturation (Figure 5.14) indicate a relatively high proportion of soil samples with high base saturation; this could be a result of the low rainfall amounts incapable of leaching the bases in the exchange soil solution. The poor leaching could also be as a result of insufficient drainage due to low soil permeability. Poor soil drainage invariably leads to high surface runoff and subsequently soil erosion. The base saturation results also show percentages higher than 100%, which is indicative of the presence of carbonate precipitates. Leaching is also inhibited by a shallow restrictive horizon that dominates within some soils in the catchment, maintaining a high base saturation. The percentage of base saturation is generally positively correlated to the soil pH. The variability in percentage base saturation is related to the position of the soils in the soil catena. The Ca:Mg ratios show that 75% of the samples have a value greater than 1, entailing higher calcium concentrations in comparison to Mg. The combined effect of Na, Mg and K is however still considerably high and induces dispersion.

The CEC values provide an indication of the quantity and type of clay present. The swelling capacity is directly linked to the CEC, as swelling increases with increasing CEC (Christidis, 1998). The CEC Clay results suggest that the clay composition in the sampled sites largely comprised 2:1 type silicate clays. The analysis indicates that illite, a mixture of illite-montimorillonite and smectites (i.e montimorillonite) are the most abundant clay minerals in this part of Keiskamma catchment.

In order to test the significance of the difference in the means of the sodium adsorption ratios of the soil A and B horizons, a student t-test was computed (Table 5.7).

Table 5.7 t-Test: Two-Sample Assuming Unequal Variance

	Soil Horizon A	*Soil Horizon B*
Mean	2.381	4.139
Variance	0.938	4.488
Observations	11	13
Hypothesized Mean Difference	0	
df	17	
t Stat	2.679	
P(T<=t) one-tail	0.008	
t Critical one-tail	1.740	
P(T<=t) two-tail	0.016	
t Critical two-tail	2.110	

The null hypothesis that there is no significant difference in sodium adsorption ratios between horizon A and B was tested. Given that the student t-test value of 2.679 is greater than the critical value (1.740) at 95% significance level, the null hypothesis is thus accepted. The sodium adsorption ratios of the B-horizon are not significantly higher than those of the A-horizon.

Particle size distribution

Particle size analysis was undertaken to determine particle size distribution. Soil texture is an important aspect that governs soil erobility. The particle size distribution results are shown in the tenary diagram (Figure 5.15) and in Table 5.8. The results show that the soil texture for most of the samples were sandy loam and sand clay loam. Sandy loam and sand clay loam soils found in this study are associated with high erodibility. This is due to high silt and very fine sand proportions which renders them easily detachable without a binding agent such as organic matter. As can be noted from the ternary diagram, the clay fraction is well above 10% for most soil samples. A clay fraction of more than 10% is adequate to supply colloids to sustain dispersive piping. The results indicate very low organic content in the soils sampled. The organic carbon percentage ranges from 0.21 to 1.02 along the hillslopes. This could be a result of the long term tillage before the land abandonment during the betterment programme which destroyed the soil organic matter. Tillage also negatively affects the soil structure.

A number of techniques have been applied in this study to land use/cover change and land degradation. A synthesis is thus required to provide a holistic overview of the results and appraisal of remote sensing and GIS techniques used on this study. This aspect in addressed in the next chapter and appropriate recommendations are given. Future directions are also given based on some gaps identified in this study.

Table 5.8 Particle size distribution and soil erodibility

Sample	% Total Sand	%Clay	%Silt	%Course Sand	%Medium Sand	%Fine Sand	% Sand <0, 1 mm	% Sand >0, 1 mm	Texture	%Organic Carbon	Soil Erodibility (K)
1	46.2	8.4	45.4	6.4	8.9	31	21.1	25.1	Lm	0.28	0.087
2	58.2	16.4	25.4	2.5	1.5	54.2	47.3	11	SaLm	0.85	0.076
3	46.2	32.4	21.4	2	5.6	38.7	30	16.3	SaClLm	0.22	0.054
4	54.2	16.4	29.4	1.1	1.6	51.5	36.9	17.3	SaLm	1.65	0.072
5	64.2	16.4	19.4	6.9	2.2	55.1	38	26.3	SaLm	0.55	0.072
6	52.2	28.4	19.4	7.7	2.3	42.3	33	19.3	SaClLm	0.28	0.058
7	76.2	10.4	13.4	4.5	5.2	66.6	42.1	34.2	SaLm	0.32	0.086
8	66.2	14.4	19.4	8.1	2.3	55.9	41.1	25.2	SaLm	0.93	0.073
9	66.2	18.4	15.4	23.8	5.8	36.6	29.3	36.9	SaLm	0.34	0.049
10	58.2	22.4	19.4	14.2	3	41.1	32.5	25.7	SaClLm	0.27	0.062
11	60.2	14.4	25.4	2.2	2.5	55.5	36.3	24	SaLm	0.55	0.081
12	54.2	20.4	25.4	6.7	1.2	46.3	35.4	18.8	SaClLm	0.86	0.071
13	76.2	10.4	13.4	1.9	6.2	68.1	44.6	31.7	SaLm	0.18	0.088
14	80.2	8.4	11.4	3.6	19.4	57.3	37.7	42.6	LmSa	0.21	0.075
15	74.2	12.4	13.4	6.6	6.3	61.3	44.8	29.4	SaLm	0.72	0.075
16	66.2	16.4	17.4	14.8	3.8	47.7	39	27.2	SaLm	1.02	0.060
17	80.2	8.4	11.4	25.3	6.5	48.5	35.4	44.9	LmSa	0.79	0.062
18	64.2	10.4	25.4	1	2.4	60.8	47.3	16.9	SaLm	0.45	0.092
19	60.2	16.4	23.4	3.1	4.8	52.4	37.2	23	SaLm	0.86	0.072
20	78.2	10.4	11.4	44.9	4.8	28.6	23.5	54.7	SaLm	0.3	0.042
21	49.6	22.4	28	4.9	1	43.7	33,2	16.5	SaClLm	0.38	0.072
22	61.6	16.4	22	5	0.8	55.9	47.5	14.1	SaLm	0.98	0.073
23	63.6	14.6	22	2.6	2	59	47.9	15.8	SaLm	0.51	0.081
24	75.6	8.4	16	10.1	4.6	60.9	44.3	31.3	SaLm	0.74	0.081

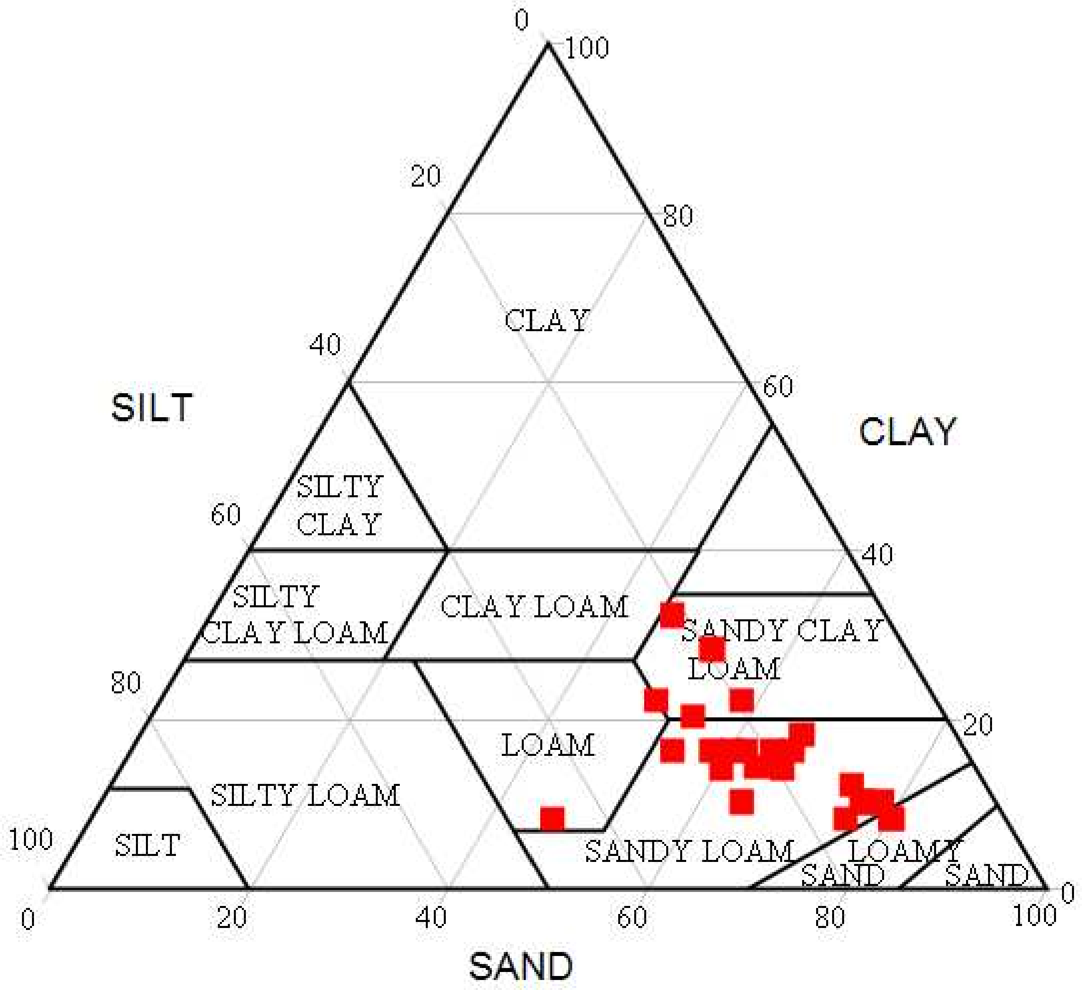


Figure 5.15 Tenary diagram showing soil texture

A general assessment of these results shows that most of the soils have high to very high erodibilities. Low organic content and high silt and fine content could be largely responsible for the high erodibilities. The organic content of all the soil samples falls below 2%, which is considered as the threshold below which soils are erodible. A linear decrease in erodibility with increasing organic content was noted by Morgan (2001) over a range of 0 to 10%. Sealing and high surface runoff is also more pronounced in soils with very low organic content.

5.3.5 Classification of erosion features and valley infill in ephemeral streams

The ability of the object oriented based multiresolution segmentation to delineate soil erosion features such as gullies is illustrated by Figure 5.16, where bright white surfaces can be separated from the other land cover types. The object oriented classification results are shown in Figure 5.18. Accuracy assessment results (Table 5.9) indicate that object oriented classification is an effective means of mapping erosional features and valley infill in ephemeral streams. An overall accuracy of 92% and kappa coefficient of 0.9 was achieved in the classification. The user's accuracy for valley infill and erosional surfaces was 93.8% and 95.3% respectively. This classification illustrates the occurrence of valley infill within ephemeral stream channels and the presence of gully erosion on the adjacent hillslopes.

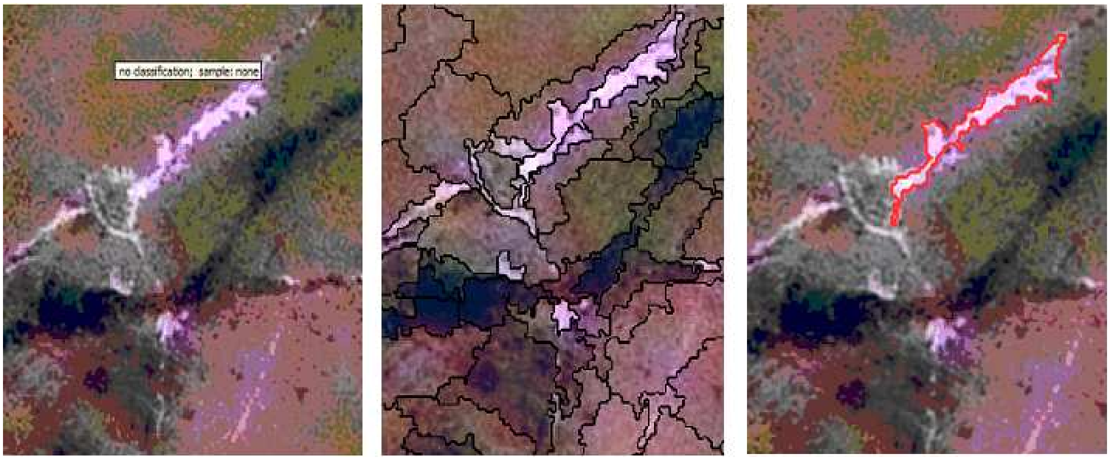


Figure 5.16 Delineation of gullies using multiresolution segmentation

Figure 5.17 Hillslope erosion and valley infill in ephemeral streams

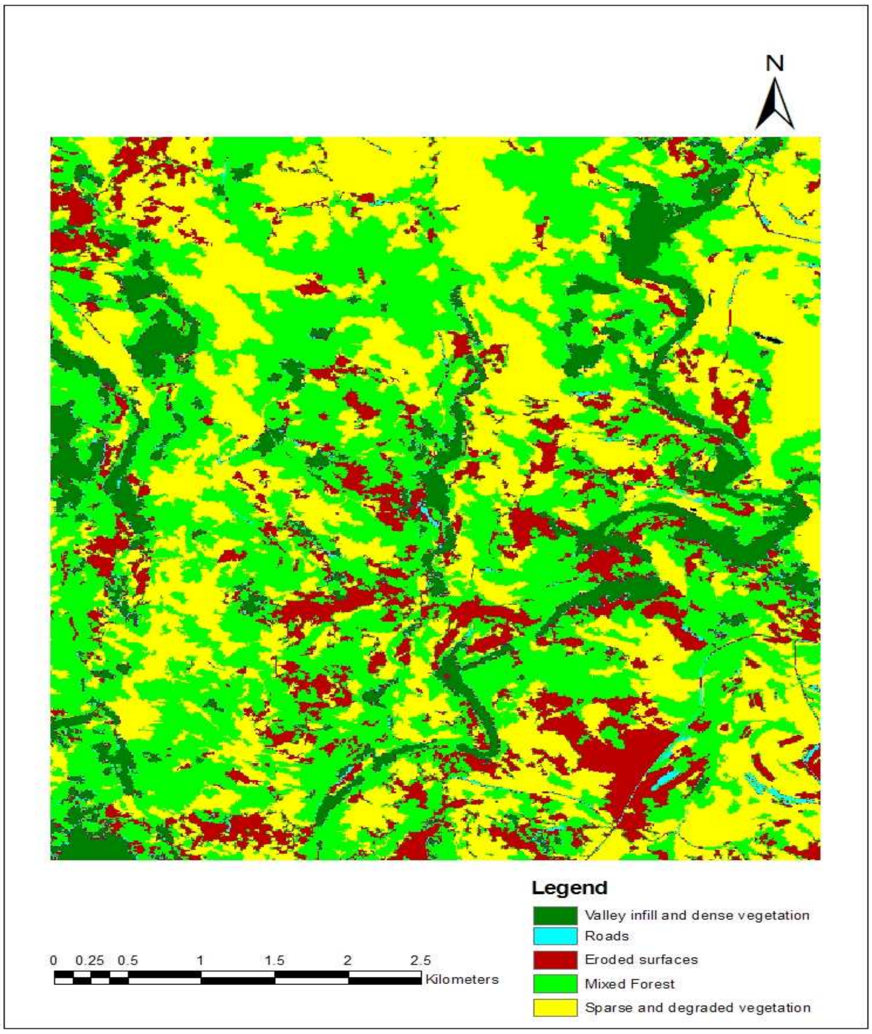


Figure 5.18 Classification of valley infill and eroded surfaces

Table 5.9 Accuracy assessment for classification of valley infill

Class	*Producer's Accuracy*	*User's Accuracy*	*Overall accuracy*	*Kappa*
1	1.000	0.938		
2	0.909	0.952		
3	0.952	0.91		
4	0.700	0.975		
5	0.950	0.905		
			0.921	0.899

1= Valley infill; 2= Roads; 3= Erosional surfaces; 4= Mixed forest; 5= Degraded Forest

These results indicate that both erosion features and sites of sediment deposition can reliably be mapped using object oriented classification.

5.4 Discussion

This chapter integrates a number of aspects that affect the soil erosion and sediment transfer processes in the catchment. The SATEEC model was used to determine the soil loss patterns in the catchment. This model integrated the principal factors affecting soil erosion which are: vegetation cover, topography, conservation practice, rainfall erosivity and soil erodibility. Topographic thresholds for gully erosion were determined using the Mahalanobis distance analysis. Areas susceptible to gully erosion were modeled using the Mahalanobis distance model. The soil physical and chemical characteristics were also examined to determine their implications for piping and gully erosion. Object-oriented classification was used to effectively map gullies and valley infill as a means of showcasing the sediment transfer processes in many parts of the catchment. The integration of GIS modeling, object-oriented classification and the physico-chemical analysis of soils provided a holistic overview of the key processes affecting soil erosion in the catchment. A discussion of each aspect examined in this chapter follows below.

The soil loss results show the Keiskamma catchment is experiencing high proportions of soil loss that are above provincial and national averages. The results indicate that the interplay between all the RUSLE factors strongly influence annual soil loss. It is noticeable that areas associated with high rates of soil loss are closely linked to communal settlements where overgrazing and wood harvesting greatly reduce vegetation, leaving the highly erodible soils vulnerable to the effects of soil erosion. Low rates of soil loss are associated with the stringent conservation practices in protected areas such as nature reserves, game parks and forest plantations. Vegetation cover in mega-conservancy areas has a significant curtailing effect on soil loss. Despite the buffering effect of vegetation in the protected zone, high rates of soil loss were noted in its peripheral areas.

Inaccuracies in the soil loss results obtained in this study are due to limitations linked to some of the parameters used in this study. Hui *et al*. (2010) indicate that USLE based models tend to overestimate soil loss due to sediment deposition on irregular and long slopes. Slope length segmentations were not accounted for in this study. The absence of the daily rainfall data (R) constrained a more accurate estimation of the soil erosivity factor. While the SDR equations developed by Vanoni (1975) provides a suitable option if catchment specific coefficients are not available, a more accurate estimation of soil loss can be obtained if catchment-specific coefficient and exponent values of SDR are used.

The application of the Mahalanobis distance model was successful in determining topographic thresholds and modeling other topographic zones susceptible to gully erosion. The preponderance of gulling in concave slopes is firstly a result of the reduction in critical shear stress in saturated soils. Secondly, convergence increases the prospect that concentrated flow will develop on the surface (Burt and Butcher, 1985; Thorne *et al*., 1986). Soil saturation plays a critical role in the development of gullies and is partially controlled by topographic wetness and planform curvature (Burt and Butcher, 1985). The results indicate the predominance of gullies in lower slopes; the reasons for this occurrence are manifold. Laker (2004) indicates that a lot of cultivation was done in the lower slopes. Recently Kakembo *et al*. (2009) revealed that seventy five percent of the gullied area was identified on abandoned lands. The scholars further indicate that topographic thresholds for gully initiation are affected by cultivation, soil structure and soil moisture. The topographic thresholds are also influenced by climate, vegetation, soil and land use (Vandekerckhove *et al*., 2000; Poesen, 2002; Kakembo *et al*., 2009). These aspects partially explain the dominance of gully erosion in lower slopes.

The results of this study also show that the soil physical and chemical properties in the Keiskamma catchment significantly contribute to the effects of piping and gully erosion witnessed in the catchment. The presence of high proportions of illite tends to promote dispersion even at low SAR values. Significant levels of smectites found in the catchment are responsible for piping and eventual gully formation. The presence of illite accounts for the highly dispersive nature of the soils even at low SAR values. Churchman *et al*. (1993) ascertain that at any given ESP, the dispersive propensity of illite predominated clay fractions is much greater than soils with other clay minerals. The presence of expanding 2:1 clays particularly montimorillonite is problematic because of their high degree of swelling and shrinkage, causing wide cracks, high degree of plasticity and subsequent proclivity to soil erosion. The dominant type of clay present in the sampled soils was estimated from the CEC ranges of the selected colloids provided by Brady and Weil (2008). The CEC for smectite (e.g. montmorillonite) ranges from 80 to 150, vermiculite 100 to 200, fine mica (e.g. illite) 10 to 40 and kaolinite 1 to 15. Humus generally contributes significantly to CEC values; however its effect in this study was low due of the low organic carbon fraction of less than 2% in all soil samples. The variation in clay mineralogy is dependent on the weathering intensity (Laker, 2004). The presence of 2:1 clays with high CEC values like smectites may be due to mild weathering occurring in the semi-arid parts of the Keiskamma catchment. The absence of kaolinite in the sampled soils could be a result of the low degrees of weathering experienced in the semi-arid areas of the catchment. Results from the study also indicate that SAR values in the A and B soil horizons surpass the threshold for sodicity and dispersion. The clay fraction is easily mobilized and easily prone to soil erosion by water.

Soil erosion in the Keiskamma catchment might be directly linked to the land use history which entailed land cultivation and abandonment in the 1960s and 1970s. The betterment programme in the 1960s witnessed extensive cultivation above the sustainable topographic slope thresholds in a lot of hillslopes in the area (D'Huyvetter, 1985). This was subsequently followed by widespread land abandonment. Long term cultivation tends to mix the thin A-horizon with the highly sodic B-horizon which increases the dispersive and crusting nature of the original A-horizon. The soils in abandoned cultivated lands also suffer from aggregate instability resulting from the effects of previous cultivation rendering them highly erodible. Structural degradation occurring in the topsoil as a result of low organic content due to the effects of previous cultivation renders the soils dispersive even under low ESP levels. The soils are predisposed to dispersion and gully erosion due to the higher sodium concentrations. The long term effects of cultivation include reduction in aggregate stability through ploughing and destruction of the soil organic content; this renders the soil more vulnerable to soil erosion. The interaction between topographic variables and soil characteristics therefore plays a critical role in gully development.

Object oriented classification was able to effectively map erosion surfaces and valley infills prevalent in many parts of the catchment. Vegetation enrichment in the ephemeral streams occurs at the expense of high soil losses from severe gully erosion on the hillslopes. Vegetation growth in ephemeral channels is promoted by enriched sediment feed from hillslopes being deposited into the channels. The highly enriched grass within the main stream channels is now a source of pasture for cattle, sheep and goats. This in turn has led to an inversion of grazing patterns within the catchment, such that grazing is now concentrated within the ephemeral stream channels. This will continue to threaten the ecological health status of the Keiskamma catchment.

Considering that various aspects have been integrated successfully in this chapter to explain the soil erosion phenomena in the Keiskamma catchment, it is evident that remote sensing and GIS methods need to be supported by soil physico-chemical analyses in order to fully understand the soil erosion processes. Object-oriented classification and GIS modeling are powerful methods for mapping soil erosion patterns and calculating soil losses. Notwithstanding this benefit, soil physical and chemical analyses are still required to provide the much needed data for computing the soil erodibility factor required in the RUSLE model and other soil erosion models. It also provides insights into the processes and inherent soil characteristics driving severe forms of erosion, such as piping and gully erosion.

5.5 Conclusion

This study reveals the spatial distribution patterns of soil loss and critical sites where erosion and deposition occur within the catchment. The study provides further evidence of alarming soil erosion rates within the catchment as a result of anthropogenic activities. The role of human activities in controlling vegetation cover and other conservation management initiatives has been noted to have either a negative and positive effect on soil erosion. This aspect is particularly demonstrated by the low soil losses in the protected areas and mega conservancy zones, and the high soil losses in other parts of the catchment such as communal areas with no effective conservation management practices in place. The removal of ground cover through thicket clearance could be curbed by introducing strong communal governance with a robust environmental management framework. Sediment delivery ratios integrated in SATEEC and object- oriented classification has effectively mapped the sediment sources and sinks in the Keiskamma catchment. The Mahalanobis distance analysis is a powerful method for computing topographic thresholds for gully erosion. The topographic threshold identified indicates that gully erosion is more prevalent in concave low lying slopes that have high topographic wetness and steam power indices. This study provides insights into typical conditions for gully occurrence within the Keiskamma catchment in terms of the potential for gully erosion. This information is essential in targeting areas vulnerable to gully erosion for consideration as high priority areas when implementing preventive environmental measures.

The soil physical analysis shows that most of the soils are sandy loams and sand clay loams with very low organic content rendering them highly erodible due to their high fine sand and silt content. Soil chemical analyses indicate that the soils are highly dispersive, promoting piping and gullying owing to the high sodium content and low soluble salt concentration. The presence of high amounts of illite-smectite in the catchment accounts for the highly dispersive nature of the soil even at low SAR values. Significant amounts of swelling 2:1 silicate clays such as smectites cause cracking and contribute to the development of gullies and pipes in the catchment. The study concludes that the physico-chemical properties of the parent material within the sediment source areas are highly erodible and they significantly contribute to piping, and gully erosion. The object-based classification was also able to effectively map the occurrence of pasture enriched valley infill flourishing in sediment laden ephemeral stream channels supplied by hillslope rills and gullies, which act as the major sediment sources. The valley infill phenomenon has given rise to an inversion in grazing patterns observed within the catchment. Grazing is now concentrated within the pasture rich ephemeral stream channel beds, as opposed to the depleted adjacent hillslopes, which are now degraded by severe rill and gully erosion. Severe degradation of the riparian zone is the inevitable outcome of this inversion. Restoration efforts should focus on rehabilitating eroded hillslopes to prevent further degradation of the riparian zone ecological status.

A number of techniques have been applied in this study to model land use/cover change and land degradation. A synthesis is thus required to provide a holistic overview of the results and appraisal of remote sensing and GIS techniques used on this study. This aspect in addressed in the next chapter and appropriate recommendations are given. Future directions are also given based on some gaps identified in this study.

Chapter 6. Synthesis

6.1 Introduction

This chapter provides an integrative review of the results obtained in this study and their implications for the catchment health of the Keiskamma catchment. The effectiveness of the techniques implemented in this study is also evaluated. A discussion of the possible interventions required to reverse increasing degradation trends in the Keiskamma catchment then follows. Future directions for research are proposed and the final conclusions of this study are drawn.

6.2 Appraisal of remote sensing, GIS and soil analysis techniques

This study demonstrates that remote sensing and GIS techniques can contribute significantly in assessing land degradation. Object oriented classification and post-classification change detection techniques were applied in this study to determine the land use/cover trends in the Keiskamma from 1972 to 2006 using Landsat satellite imagery. Object oriented classification marks a significant paradigm shift in remote sensing image classification since traditional remote sensing image classification was based on per-pixel classification algorithms. The use of object oriented classification techniques in degradation assessment has not yet gathered momentum in South Africa. This study therefore demonstrates the feasibility of using object oriented classification in assessing land cover trends and gully erosion mapping. Besides the high classification accuracy achieved in using object oriented classification in this study, image objects derived from multiresolution segmentation closely reflect ecological units or patches. This presents significant advantages in change detection studies involving landscape patterns and structure. This study shows that multiresolution segmentation can produce meaningful and tangible landscape objects that can be regarded as patches which are relatively homogenous to which one can apply fragmentation analysis to assess change. Image objects contain more attributes compared to single image pixels, which makes object oriented classification more competitive. The mixed pixel effect which is eliminated in object oriented classification makes the classification more suitable for change detection analysis. The results from this study show that object oriented classification provides a fast and effective way for post-classification change detection, and thus provides an alternative means to change detection using per-pixel classification.

The weakness of the object oriented post-classification change detection approach used in this study is that it does not allow detection of subtle and minor changes within land use/cover classes. While this setback also confronts per-pixel post-classification change detection (Rogan et al., 2003), it is magnified in object-based methods since similar pixels are amalgamated to form objects. This diminishes the likelihood of detecting subtle changes within land cover classes.

Although high classification accuracies were achieved in this study, they were still subject to a number of errors as shown by the error matrix (Appendix A). Shao and Wu (2008) indicate that these errors can be propagated into landscape pattern analysis and post-classification change detection. Furthermore whilst this study attempted to resolve issues regarding differences in spatial resolution between Landsat MSS and Landsat TM to insignificant levels at regional scale, the methods may not completely remove the differences in temporal resolution or grain sizes (Wu, 2004). It is also evident from the error matrix (Appendix A) that whilst the overall accuracies are high, the discrimination between settlements and bare and degraded soils was difficult resulting in lower class accuracies for these classes, particularly with Landsat MSS imagery. This could be attributed to the spectral similarity between these two classes in communal areas and the low spatial resolution of Landsat MSS imagery. Although Wickham *et al*. (1997) indicate that bias in landscape metrics are not significantly amplified by land cover misclassifications, the high accuracies obtained in this study could still be improved by using high resolution imagery. Whereas the accuracies of object-oriented classifiers and advanced per-pixel classifiers may be comparable at medium spatial resolution, object oriented classified images are more suitable for landscape analysis (Clark and Pellikka, 2009).

Gully mapping using per-pixel classification is generally problematic; this study demonstrates the efficacy of hierarachical object based classification in mapping rill and gully erosion and valley infill using pansharped imagery. The high success rates achieved in mapping gullies and valley infill are based on factors inherent in Definiens object oriented classification software. The study also demonstrates the usefulness of fused multispectral Landsat 5 TM and panchromatic SPOT 5 data in the mapping of sediment sources and valley infill using object oriented classification. Furthermore, the Principal Components pansharpening algorithm has also proved to be an effective pan sharpening algorithm that enhances both the spatial and spectral resolution satellite imagery. The ability of multiresolution segmentation to delineate gullies enables efffective rule based classification using spectral, brightness, geometrical features.

The integration of fragmentation analysis done in the assessment of land use/cover change provided several advantages in the inference of landscape patterns. Fragmentation analysis executed in the communal areas of the central Keiskamma, and in the riparian and proximal hillslopes was effective in establishing the link between landscape patterns and processes occurring within these areas. While many studies in South Africa have focused on time series using satellite imagery, this study extends the analysis of land use/cover change by using landscape metrics to infer anthropogenic and hydrological processes impacting on land use/cover change. The use of patches and other ecological units derived using object base classification in the analysis of landscape structure and fragmentation was considered to be more meaningful and effective in the assessment of change than per-pixel classified geographical entities. Esbah *et al*. (2010) concurs that object-based techniques incorporate spatial neighbourhood properties into the classification process that result in a more accurate representation of landscape patterns compared to pixel-based methods. The landscape metrics used in this study also effectively described the landscape condition and long term responses to anthropogenic impacts in the catchment. For instance, the fragmentation taking place in riparian and proximal hillslopes was effectively captured by the fragmentation analysis using FRAGSTATS as validated in the field. Anthropogenic processes such as deforestation, cultivation, overgrazing were manifest. Fragmentation processes were revealed by smaller patches of intact vegetation which were more isolated. The incorporation of landscape metrics in the land use/cover analysis using satellite imagery thus permitted a better understanding of both the processes and land use/cover changes taking place in the catchment.

Remote sensing results often need to be validated by rapid field techniques; this study demonstrates the feasibility of integrating landscape function analysis in assessing the current landscape condition. The landscape organisation index was useful in distinguishing the levels of land degradation in the different communal villages and the former commercial farms. Landscape function analysis proved valuable in quantitatively verifying information derived from satellite imagery. In the same vein, the rapid assessment of riparian condition can be used to assess the condition and fragmentation in riparian and proximal hillslopes in support of remote sensing techniques.

The feasibility of using Markov-Cellular Automata simulation model in land use/cover projection provides a critical contribution to conservation and land use planning in the catchment. Considering the increasing land degradation trends in the Keiskamma, the simulated future land use/cover map can provide an effective indicator of possible future scenarios, if the current rates of land cover change persist. The downside of the Markov Cellular Automata as executed in this study is that the effect of climate change and other climatic events such as drought and floods has not been considered. These factors have considerable potential to affect the predicted scenarios. That notwithstanding, the Markov Cellular Automata model however, still provides useful future indicators for sustainable planning and has been adopted in many parts of the world (Wood *et al*., 1997; Petit *et al*., 2001; Weng, 2002; Gómez-Mendoza *et al*., 2006; Ye and Bai, 2008; Guo *et al*., 2009; Kamusoko *et al*., 2009). The implementation of the Markov Cellular Automata in this study basically answers the call by Scogings and Lent (2000) for predictive methods for land use/cover dynamics in the Eastern Cape. This study demonstrates the possibilities of adopting a proactive environmental management approach rather than a reactive tendency evident in most parts of Southern Africa.

The SATEEC model integrated in GIS, proved to be a useful tool in rapidly assessing soil erosion potential at catchment scale. SATEEC model is suited for use at catchment scale since it is incorporated with algorithms to compute sediment delivery ratios. While the result of the SATEEC model is acceptable as shown by the field validation, some inaccuracies are still present. Many scholars point out that models developed from USLE tend to overestimate soil erosion and sediment yield due to sediment deposition on irregular and long slopes (Hui *et al*., 2010). This is because of the limitation of the slope length factor that does not account for the slope length segmentations in the catchment (Kang *et al*., 2009). Future studies should address this weakness by including a roads and contour layer in the modelling process. In cases where the daily rainfall data (R) are available, the daily R factor can be used to estimate sediment yield more accurately (Woo *et al*., 2010). While the SDR curve developed by Vanoni (1975) was the best alternative in this study, SDR equations derived from catchment-specific coefficient and exponent values are required to improve the accuracy of the sediment yield estimation.

The Mahalanobis distance analysis traditionally used in wildlife habitat ecology has been used successfully to determine topographic thresholds susceptible to piping and gully erosion. The application of the Mahalanobis distance method to determine topographic thresholds susceptible to severe forms of erosion has not been explored in local studies.

The characterization of the soil physical and chemical properties provided critical insights into processes driving piping and gully erosion. Laboratory methods were useful in determining the intrinsic soil physical and chemical affecting soil erodibility particularly dispersion. The functional relationships between physico-chemical parameters such as EC and ESP, % clays and SAR, EC and % Base saturation, pH and SAR was useful in determining the dispersive nature of the soils in the Keiskamma catchment. The analysis of particle size distribution has proved to be useful in providing data for calculating soil erodibility, a factor which is required in the RUSLE model. GIS models are data driven, and soil analysis provides a critical input to a number of GIS models besides SATEEC used in this study. A call for the inclusion of the physical and chemical characterization of soils in soil erosion studies was made by Laker (2004) in a review of soil erosion studies done in South Africa. The importance of characterizing the physical and chemical properties of soil in areas to prone to piping and gully erosion is highlighted in numerous studies (Faulkner *et al*., 2000; Jones *et al*., 2010; De Santis *et al*., 2010; Verachtert *et al.*, 2010).

6.3 A review of the results

6.3.1 Temporal vegetation change and fragmentation analysis results

The results of the temporal analysis done using the post-classification change detection indicate that intact vegetation has undergone a significant decline from 1972 to 2006. The temporal changes within the intermediate years are characterized by cyclic transitions of decline and recovery of intact vegetation. On the whole, results indicate a decline in intact vegetation cover, an increase in degraded vegetation and bare eroded soil. Fragmentation analyses done in the communal villages of the central Keiskamma catchment indicate increasing vegetation fragmentation manifested by an increase in smaller and less connected vegetation patches and a subsequent increase in bare and degraded soil patches which are much bigger and more connected. The differences in rangeland condition in the different communal areas and the former commercial farms were validated using the LOI. The LOI revealed very low vegetation connectivity in communal rangelands that have weak local traditional institutions. In contrast, good range conditions existed in communal rangelands with strong local institutions. These differences were investigated by Bennett and Barrett (2007) who suggested that the differences in rangeland condition are a reflection on the degree of control local communities exert on communal grazing resources. Their study reveals that grazing resources are influenced by the social and ecological heterogeneity that characterise the catchment (Bennett and Barrett, 2007). Moyo *et al*. (2008) concur that the rangeland conditions and grazing strategies found in the communal areas are a sequence of the interaction between social, land tenure, ecological and institutional factors. In a similar study, Ainslie (2002) suggests that dissimilarities in rangeland condition in communal grazing areas are a result of high stocking density and ineffective rangeland management methods. The strength of local institutions such as Residence Associations (RA) and traditional authorities responsible for coordinating grazing and land management in communal villages explain the disparities in rangeland condition in the central Keiskamma (Bennett and Barrett, 2007; Moyo *et al*., 2008). This study also confirms that former commercial farms have better rangeland condition compared to the communal areas; this is proved by higher image analyses and landscape indices which both reflect relatively high vegetation connectivity. The differences are however not significantly different from those of communal villages whose rangeland condition is still good.

Fragmentation analyses in the riparian and proximal hillslopes revealed evidence of increasing vegetation fragmentation from 1972 to 2006. This is borne out by the reduction in intact vegetation and subsequent increase in bare and degraded patches. Fragmentation in the riverine and proximal hillslopes is directly caused by a number of anthropogenic factors such deforestation, overgrazing and cultivation. A reduction in intact vegetation and an increase in degraded vegetation species could be linked to permanent loss of saturation induced by river impoundments. Nilsson and Berggren (2000) suggested that a reduction in the plant water requirements could stress native vegetation and lead to the successive dieback of older plants while inhibiting regeneration of younger pioneer species. The scholars also ascertain that disturbances in the riparian zones open ways for invasion by exotic species. The occurrence of xeric and other invasive plant species within the proximal hillslopes which indicate soil moisture losses that could be related to the long term post impoundment hydrological changes. This inference however, warrants further investigations. Evidence of significant fragmentation in riparian and proximal hillslopes was collected using rapid assessment of the riparian zone.

6.3.2 Markov Cellular Automata predictions

The Markov Cellular Automata simulation results predict a reduction in intact vegetation and an increase in bare and degraded soil in 2019. The predicted scenario implies an increase in land degradation. Marginal increases are predicted for degraded vegetation and in this simulation degraded vegetation species may be acting as efficient catalysts that transform intact vegetation to bare surface connectivity.

6.3.3 Soil loss results

The SATEEC computations from this study indicate a mean soil loss of 36.063 tons/hectare/year. This rate of soil loss is very high considering that average soil loss reported by Le Roux *et al*. (2008) for South Africa is about 12 tons/hectare/year. McPhee and Smithen (1984) proposed soil loss tolerances from 3 tons/hectare/year for shallow soils and 10 tons/hectare/year for deep alluvial soils in South Africa. It is clear that the Keiskamma catchment is currently experiencing excessive rates of soil losses way above sustainable tolerance limits. The role of human activities in such high soil losses is imprinted in various ways. Protected and megaconservancy zones are characterized by low soil losses while communal areas with weak local institutional structures to govern rangelands have very high soil losses. The results indicate that 47% of Keiskamma catchment has soil losses higher than 12 tonnes/ha/yr. It can be concluded that the rates of soil loss in the Keiskamma catchment are excessive and are above the provincial and national averages. The SATEEC results effectively illustrated the spatial distribution of soil loss throughout the Keiskamma catchment. This occurrence is widespread throughout the Keiskamma catchment and has strong implications for grazing patterns and land degradation. Grazing is now shifted to the ephemeral stream channels, which further exercebates the riparian zone condition.

6.3.4 Topographic thresholds for gully erosion results

The topographic thresholds identified for severe forms of erosion in the Keiskamma catchment showed gully erosion is predominant in the lower slopes. The results indicate that gully erosion occurs predominantly in slope angles with a mean vector of 6.83764°. This value is below the average slope of 8.497° for the Keiskamma catchment. This indicates that lower slopes are more vulnerable to gulling and that most gullies occur in areas of high topographic wetness with a mean vector of 4.86929. This is higher than the average topographic wetness of 4.041 for the catchment. Gully erosion is also dominant in areas of higher Stream Power Index (SPI) where mean vector value for gully occurrence is 1.06333; yet the SPI for the catchment is 0.723. The prevalence of gullies on concave slopes is notable, as depicted by a mean vector of -0.06968 for the planform curvature. The topographic slope angle and position, wetness index, and SPI play a significant role in the distribution of gully erosion in the catchment. This study also demonstrates that the Mahalanobis distance analysis integrated in GIS is a useful method to determine topographic thresholds for gully erosion.

6.3.5 Soil physical and chemical characterization

The soil physico-chemical characterizations reveal that the soils in the Keiskamma comprise dispersive 2:1 clays that are highly erodible. It is concluded that intrinsic soils properties significantly contribute to the development of piping and gully erosion in the catchment. The predominance of illite tends to promote dispersion even at low SAR values. Significant levels of smectites in the soils account for piping and eventual gully formation. The SAR values in the A and B soil horizons exceed the threshold for sodicity and dispersion, thus significantly accelerating severe forms of erosion. The significance of the intrinsic soil physical and chemical characteristics in piping and gully erosion was also confirmed by many scholars (Faulkner *et al*., 2000; Faulkner *et al*., 2003; De Santis *et al*., 2010; Jones, 2010; Verachtert *et al.*, 2010).

6.4 Recommendations

Field observations indicate that rehabilitation measures have been implemented before in the Keiskamma catchment but have done very little to curb land degradation. These include poorly designed contours, stone barricades to curb gully erosion and the planting of sisal reeds around gullies. The failure of these methods to curb erosion could be attributed to lack of regular follow up action. Reversing the environmental degradation patterns in Keiskamma catchment requires a holistic approach which integrates community regulatory frameworks, legislation, stewardship, together with socio-economic transformations. Thus the success of the rehabilitation programmes requires many actors such as Government, Non- Governmental Organisations (NGO) and the local communities.

In the light of the high soil losses and deteriorating vegetation condition in the communal areas, resettlement is proposed as a possible means of alleviating anthropogenic pressure in the communal areas. This could be achieved through a systematic re-allocation of state land in sections of the catchment that belonged to the former commercial farms. De Wet (1994) supports resettlement as a possible means to alleviate land degradation in communal areas and presents a number of possible settlement patterns and related problems. In addition, Cousins (2007) proposed that the agrarian issue should be resolved through a wide-ranging agrarian reform that restructures rural economic space, property regimes and socio-political relations in addition to land distribution and granting of land rights. This proposition finds relevance in land degradation because of the multiple livelihood prospects which emanate from the expanded opportunities availed by this strategy. It is pertinent however that in dealing with policy issues related to land degradation, land-use intensification and erosion, resettlement alone might not necessarily be the key. To ensure that resettlement succeeds, Fox and Rowntree (2002) suggested that it is crucial to evaluate the land potential for such designations in terms of biological productivity, terrain classification, degradedness, agro-climatic zonation and land tenure. While resettlement remains a viable and practical measure to ease anthropogenic pressure on the environment, it is not in itself a panacea to reverse degradation trends since new areas could still be subjected to the same socio-economic dynamics driving land degradation in the communal areas.

Considering that communal rangeland conditions are a reflection of the strength of local traditional institutions, the role of local institutions in managing rangelands and grazing systems should be reinforced. This aspect was studied by Bennet (2008) who examined the constraints of rangeland management as a common property in Central Eastern Cape and concluded that open-access approaches contribute significantly to land degradation. Moyo *et al*. (2008) highlight that lack of strong local-level institutions, little knowledge of veld management, nonexistence of rules and absence of seasonal restrictions on rangeland resources are some of constraints to effective rangeland management. Open-access rangeland management is impeded by failure to define and enforce rights to particular grazing resources, coupled with fragile local institutions incompetent of managing rangelands effectively. In view of the decreasing vegetation cover and excessive soil erosion in communal rangelands, rotating the grazing camps is recommended in communities with a weak communal governance system. The system of rotational grazing camps is not a new concept in the communal areas of the Eastern Cape, all which is required is its enforcement in all the communities. Fences and paddocks can be used to effectively manage grazing rotations.

Controlling the high rates of soil loss and the declining vegetation condition in the Keiskamma requires effective co-ordination among the diverse resource users and stakeholders. Collaborative management or co-management is recommended as possible solution to integrate the different stakeholders in the Keiskamma catchment. A conceptual framework for co-management is provided by Plummer and Fitzgibbon (2004). Power, rights and responsibilities for natural resource management are shared between local users and government. Co-management connects government based systems and local-level systems. Collaborative management is therefore recommended suitable environmental management model that integrates the diverse stakeholders such as local tradition institutions and government departments.

To prevent the land degradation scenario predicted using the Markov model, it is necessary to educate and empower the communities of the need for vegetation and soil conservation. Environmental awareness campaigns should be done at grassroots level to ensure sufficient support from the residents. This can be done by means of gender balanced community workshops and training. Such programmes should be directed at empowering the communities about the causes and effects of soil erosion so that community designed solutions are generated to curb the effects of land degradation. Educational programmes on rehabilitation should integrate the indigenous knowledge systems of the local communities; this approach ensures that relevant local knowledge on the environment is preserved. Agricultural extension officers can play a critical role in educating the communities on issues of conservation. Communal dwellers also tend to identify themselves easily with their traditional values which they can easily follow. Participatory approaches and intra-community exchanges are feasible ways to ensure community support in environmental conservation and brain storming community based solutions to curb degradation. Systematic conservation planning should include traditional ecological knowledge. The role of indigenous knowledge should be integrated into the environmental conservation plans of the Keiskamma catchment because local people can provide valuable contributions based on their traditional knowledge, practice and belief systems developed over decades.

In view of the fact that vegetation and soils form the basic resources of the people in the Keiskamma, a Community-Based Natural Resource Management (CBNRM) is recommended as a practical means of resolving some of the environmental problems occurring in the Keiskamma catchment. While the results of this study show increasing vegetation fragmentation and high soil loss, these occurrences are driven by legitimate human needs such as fire wood collection. The CBNRM advocates that the right to control resources should be entrusted in the local communities whose livelihoods directly depend on the resources and thus have more interest in sustainable use and management of the resources than the government or external organisations. It has been applied in many parts of Africa and is a favourable model for many international funding institutions (Blaikie, 2006). The communal areas management programme for indigenous resources (CAMPFIRE) in Zimbabwe is a classical example of CBNRM. CBNRM programmes are targeted towards poverty alleviation by empowering local communities to manage resources for long-term social, economic and ecological benefits (Agrawal, 2001; Blaikie, 2006). Strategies involved in CBNRM include introduction of locally adapted conservation methods that have sound ecological principles while promoting community participation and enhancing capacity for natural resource management. It is envisaged that the CBNRM suggested for the Keiskamma will yield economic, social and ecological benefits to the rural communities and achieve environmental sustainability.

The success of CBNRM is dependent on a number of socio-economic considerations since land degradation in the Keiskamma catchment is driven by legitimate socio-economic needs to sustain livelihoods. Hoffman and Todd (2000) observed that in South Africa land degradation was closely related to poverty levels in the communal areas. Environmental rehabilitation projects in the Keiskamma catchment should therefore integrate sustainable livelihoods concepts which improve community resource management whilst addressing poverty alleviation, capacity development and project sustainability. Land rehabilitation strategies should integrate alternative means to support sustainable rural livelihoods that increase food security and meet the daily needs of the people. Livelihood programmes should promote alternative protein sources which have minimum impact on the rangeland such as poultry and piggery projects. Organic farming projects such as home vegetable and herbal gardens are an alternative. Herbal gardens could be used to minimize the effects of vegetation destruction for medicinal purposes.

Given the reduction in intact vegetation cover observed in the communal villages, rehabilitation should focus on alternatives for increasing vegetation cover within the villages. The planting of vegetation species that bring economic benefits to the local communities is therefore recommended as a means of sustaining livelihoods in the catchment whilst promoting vegetation recovery. Communities could benefit from planting eucalyptus woodlots where economic remuneration can be derived by selling poles and timber. Planting trees that could be used for bio-fuels such as Jatropha is a further example. Drought resistant cactus plants which bear edible prickly pears could also be grown is some gullied areas for human consumption. Planting of grasses which could be used to thatching is a feasible means of generating income. Homesteads should be encouraged to plant orchards and other trees that increase ground cover around their homes. Planting grasses such as kikuyu around their homesteads is recommended since they can be used as lawns. Livelihoods in the communal areas can also be improved by introducing commercial honey projects. The success of honey projects depends on improved forage; this might act as an incentive to the people to maintain vegetated areas. A shift on the dependence on vegetation should be promoted through the use renewable energy sources as alternatives to firewood. Examples include the use of solar, wind and biogas.

Looking into the future, the Markov Cellular Automata simulation results predict a decline in intact vegetation and subsequent increase in bare and degraded soil patches. Sustainable solutions which increase vegetation cover and curtail soil erosion should be integrated in the recovery plan for the Keiskamma. The Land-use and Livelihood Project initiated in the Fish-Kowie Mega Conservancy Network as part of the Fish River Biodiversity Initiative provides a useful baseline for sustainable conservation and rehabilitation that could be adopted in the Keiskamma catchment. Knight and Cowling (2006) developed a five step process summarized as:

- ✓ Formulation of a co-operatively developed common vision,
- ✓ Provision of an ecological model for sustainable land management,
- ✓ Identification of spatially-explicit conservation priority areas,
- ✓ Provision of an implementation plan
- ✓ Empowerment of individuals and institutions.

The authors highlighted that systematic conservation assessment should comprise four connected elements namely landscape values, people, institutions and instruments. This method provides a holistic approach that surpasses technical and scientific interventions alone, and warrants adoption in the Keiskamma catchment.

Taking into account the increase in degraded vegetation noticed in the temporal land cover analysis, the eradication of undesired vegetation species should be considered. The encroachment by karroid shrubs and other invasive alien species is evident in most communal areas in the Keiskamma catchment. The use of controlled intense veld fires to eradicate undesired vegetation species is recommended. Trollope (1980) provides several benefits on the use fire in managing invasive plant species.

Given the increases in vegetation fragmentation in the communal areas and excessive rates of soil loss, fast growing plants such as spekboom (*Portulacaria afra*) provide a quick solution to replace vegetation cover and reduce the rates of soil erosion. Powell *et al*. (2006) advocates for the use *Portulacaria afra* in rehabilitating the thicket because it fixes large amounts of carbon in both soils and biomass at landscape scale. A further advantage of spekboom is that it can easily be grown from cuttings. Powell *et al*. (2006) proposed that an instant bush structure could be created by planting *Portulacaria afra* truncheons, Aloe, Euphorbia, Crassula and other succulent species. The researchers predicted this could act as catalyst for a speedy recovery of the Subtropical Thicket.

The topographical thresholds identified using the Mahalanobis distance analysis provide a useful basis to control gully erosion. Topographic threshold analyses show that concave low lying slopes that have high topographic wetness and steam power indices are predisposed to gulling. Areas susceptible to gullying have also been determined using Mahalanobis distance model. It is thus recommended that special focus should be placed on the identified topographic zones. One approach is to ensure that bare low lying concave slopes are vegetated by growing plants such as Vetiver grass (*Vetiveria zizanoides*), which provide instant vegetation cover that shields the soil. Grazing should also be shifted from these areas as it tends to increase the risk of gully erosion. Contours and runoff channels should also be constructed to minimize excessive overland flow in these vulnerable areas.

The fragmentation analysis in the riparian and proximal hillslopes showed a decline in intact vegetation. The fragmentation was directly linked to anthropogenic impacts such as thicket clearing, overgrazing and cultivation. It was also inferred that the decline in intact vegetation could linked to loss of saturation arising from the long term impacts of impoundments on the Keiskamma River. Upon this basis, it is recommended that the flow regimes in the Keiskamma River should be well regulated by timing the water releases from the Sandile dam in order to maintain the plant water requirements. Deforestation and overgrazing in the riparian and proximal hillslopes should also be prohibited to ensure vegetation recovery in this critical ecological zone. Vetiver grass can still be used to stabilise areas affected by severe forms of erosion in the riparian zones and proximal hillslopes. This recommendation is based on the past successes of Vetiver grass in the Bathurst region of the Eastern Cape Province (Matoti, 1999). The Vetiver grass is sterile and thus incapable of invasive tendencies. The bamboo like plant has no grazing potential due to its hard lignified walls. Hill (1996) indicates that Vetiver grass forms dense stands similar to sugar cane plantations and its deep roots promote high infiltration and stabilizes the soil.

Given the dispersive nature of soils in the Keiskamma catchment, the use of soil conditioners and stablizers is recommended to prevent soil erosion due to clay dispersion. The application of gysum will improve aggregate stability, control soil sealing and improve infiltration. Phospogysum is also an effective universal agent for stabilizing dispersive soils. The spraying organic polymer soils in areas vulnerable to erosion before the rain season should also be considered. Levy (1995) provides a useful review of the use of soil conditioners and stabilizers.

This study recommends that policies which regulate the use of rangelands and riparian zones should be enforced to reduce land degradation. A policy which regulates the water flow regimes in rivers affected by impoundments should be enacted. Such a policy will ensure that the plant water requirements for vegetation in riparian and proximal hillslopes are met. Regulations which curb deforestation should be enforced as a measure to reduce vegetation fragmentation. Given the increasing land degradation noticed in the communal rangelands, destocking and rotational grazing polices are recommended. Such a policy will reduce pressure on the rangelands which are currently overgrazed. A land resettlement policy should also be enacted to ease anthropogenic pressure in highly degraded villages. Measures to curb environmental degradation in the Keiskamma catchment should encompass suitable ecological interventions that are sensitive to the socio-economic challenges facing the people in communal areas**.** Monitoring programmes to check the performance of the rehabilitation strategies will ensure timely corrective action is taken to avert failure of remedial initiatives. This study recommends that a holistic and participatory approach that is ecologically sustainable should be undertaken to reverse the ever increasing degradation trends in the Keiskamma catchment.

6.5 Directions for future research

Future research should focus on

- ✓ The application of radar differential interferometry and LiDAR altimetry for the generation of high resolution DEM and DTM required in the temporal analysis of gully erosion and sedimentation. These techniques are useful in the determination of the spatial distribution and magnitude of erosion in terms of volumes of soil loss. These methodologies could be integrated with the use of high resolution optical satellite imagery.
- ✓ Continuous development and refinements of models to predict future land cover/use states. Land use/cover simulation models may be improved by incorporating climatic trends and time series satellite imagery.
- ✓ Monitoring the effectiveness of environmental rehabilitation methods.
- ✓ Investigating the ecological function of degraded vegetation particularly invasive species.

- ✓ Study the correlation between rainfall trends and vegetation change in the catchment.
- ✓ Determining the socio-economic impacts of land degradation in the catchment.
- ✓ Mapping loss of agricultural land due to land degradation

6.6 Conclusion

This study makes a significant contribution to the current knowledge of the environment in the Keiskamma catchment. Land use/cover change detection analysis done using object-oriented classification point to increasing levels of land degradation. The study shows that object-oriented remote sensing is a viable method for land use/cover change detection. The conceptualization of vegetation patches as image objects was valuable in determining landscape changes using fragmentation analysis Fragmentation analyses in the riparian and adjacent hillslopes zone is manifested by a reduction in intact vegetation and subsequent increases in degraded vegetation and bare soils. The loss of intact vegetation, increased degraded vegetation and bare soil surfaces in the riparian zone and adjacent hillslopes was attributed to loss of saturation caused by the long term impacts of impoundments on riparian and hillslope vegetation. Anthropogenic impacts such as overgrazing, cultivation, deforestation and settlement also contributed to fragmentation in the riparian and adjacent hillslope zones. The fragmentation analysis in the communal areas showed evidence of increasing fragmentation since 1972. Rangeland condition in communal villages showed remarkable contrasts, with some villages exhibiting better rangeland condition compared to others. This was confirmed by the landscape function analysis. These differences were attributed to the role of local traditional institutions in governing communal rangelands. The Markov-Cellular Automata modelling results indicated significant increases in bare and degraded soils and a reduction in intact vegetation in 2019. The application of the SATEEC model was effective in modelling the soil loss patterns in the catchment, with soil loss rates far above the country's average. A rule based object-oriented classification was also successfully used to classify eroded surfaces which act as sediment sources and valley infill which are the major depositional zones. This classification demonstrates the sediment transfer processes in the catchment. The topographic thresholds computed using Mahalanobis distance analysis provided useful insights into areas with high potential for gully erosion. The study has demonstrated the effectiveness of remote sensing and GIS techniques in determining and predicting soil loss and land use/cover trends. It further provides recommendations for rehabilitation. In summary this study makes the following contributions.

1. Identifies the trends in land use/cover including the landscape changes in the catchment.
2. Predicts the future land use/cover.
3. Determines the soil loss patterns in the Keiskamma catchment.
4. Identifies the sediment source areas and depositional environments.
5. Establishes topographic thresholds of areas susceptible to gully erosion.
6. Characterizes the soil physical and chemical properties affecting severe erosion forms
7. Provides suitable recommendations to rehabilitate degraded areas.

The contributions outlined above are important in understanding the degradation patterns in the Keiskamma catchment and provide indicators of the environmental health of the catchment. The study strengthens the current understanding of the dynamics and mechanisms which characterize land degradation in the Eastern Cape. It concludes that the Keiskamma catchment is severely degraded and shows increasing trends of degradation. Urgent rehabilitation measures are required to reverse the increasing degradation predicted for 2019.

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APPENDICES

Appendix A: Accuracy assessment error matrices

1972 Error matrix based on a TTA mask

User \ Reference Class	V	W	DV	BDS	S	Sum
V	18962	0	1174	268	448	20852
W	0	414	0	0	0	414
DV	117	4	5149	811	188	6269
BDS	0	0	359	3278	747	4384
S	0	0	0	537	1022	1559
unclassified	0	0	0	0	0	0
Sum	19079	418	6682	4894	2405	
Producer	0.994	0.990	0.771	0.670	0.425	
User	0.909	1	0.821	0.748	0.656	
Hellden	0.950	0.995	0.795	0.707	0.516	
Overall Accuracy	0.861					
KIA	0.762					

1988 Error matrix based on a TTA mask

User \ Reference Class	V	W	DV	BDS	S	Sum
V	11112	314	9	299	300	12034
W	9	2596	14	0	0	2619
DV	382	5	5274	1076	91	6828
BDS	18	0	565	1959	592	3134
S	23	0	5	986	1325	2339
unclassified	0	14	0	186	42	242
Sum	11544	2929	5867	4506	2350	
Producer	0.963	0.886	0.899	0.435	0.564	
User	0.923	0.991	0.772	0.625	0.566	
Hellden	0.943	0.936	0.831	0.513	0.565	
Overall Accuracy	0.819					
KIA	0.749					

1993 Error matrix based on a TTA mask

User \ Reference Class	V	W	DV	BDS	S	Sum
V	12517	0	966	0	235	13718
W	0	5371	38	2	18	5429
DV	332	0	5441	797	523	7093
BDS	30	81	67	3052	62	3292
S	0	0	41	85	1049	1175
unclassified	0	0	0	0	0	0
Sum	12879	5452	6553	3936	1887	
Producer	0.972	0.985	0.830	0.775	0.556	
User	0.912	0.989	0.767	0.927	0.893	
Hellden	0.941	0.987	0.797	0.844	0.685	
Overall Accuracy	0.893					
KIA	0.851					

1997 Error matrix based on a TTA mask

User \ Reference Class	V	W	DV	BDS	S	Sum
V	11691	0	46	0	180	11917
W	8	5831	0	0	0	5839
DV	9	3	7046	2125	538	9721
BDS	0	22	280	6052	183	6537
S	0	0	8	112	982	1102
unclassified	0	20	0	0	0	20
Sum	11708	5876	7380	8289	1883	
Producer	0.999	0.992	0.955	0.730	0.522	
User	0.981	0.999	0.725	0.926	0.891	
Hellden	0.990	0.995	0.824	0.816	0.658	
Overall Accuracy	0.899					
KIA	0.867					

2001 Error matrix based on a TTA mask

User \ Reference Class	V	W	DV	BDS	S	Sum
V	15327	0	93	0	47	15467
W	47	4260	38	4	14	4363
DV	0	1	8191	125	598	8915
BDS	0	0	694	5901	372	6967
S	0	0	1	527	2010	2538
unclassified	2	393	0	38	108	541
Sum	15376	4654	9017	6595	3149	
Producer	0.997	0.915	0.908	0.895	0.638	
User	0.991	0.976	0.919	0.847	0.792	
Hellden	0.994	0.945	0.914	0.870	0.707	
Overall Accuracy	0.920					
KIA	0.892					

2006 Error matrix based on a TTA Mask

User \ Reference Class	V	W	DV	S	BDS	Sum
V	11356	0	0	0	0	11356
W	0	3982	0	0	0	3982
DV	354	25	7034	96	52	7561
S	0	0	77	2336	0	2413
BDS	0	0	2065	461	6317	8843
unclassified	168	210	0	0	0	378
Sum	11878	4217	9176	2893	6369	
Producer	0.956	0.944	0.767	0.807	0.992	
User	1	1	0.930	0.968	0.714	
Hellden	0.978	0.971	0.841	0.881	0.831	
Overall Accuracy	0.898					
KIA	0.867					

Appendix B: Landscape organization data

Transect No:1
Rangeland: CF
LOI: 0.558

Patch	Interpatch	Patch	Interpatch
	6		1
4.12		3.6	
	1		1
0.7		8.8	
	1		1
0.6		3.4	
	2		2.5
0.8		3.6	
	1		3
1.8		2.6	
	6		0.5
1.7		3	
	4		2
4.9		2.1	
	4		2
2.8		10	
	1		0.5
3.4		7.9	
	5		2
3.3		0.8	
	9		0.5
1.6		3.4	
	2		2
1.3		6.7	
	2		5
1.2		6.3	
	8		5
2.2			
	2		
1.6			
	3		
2.2			

Transect No:2
Rangeland: CF
LOI: 0.319

Patch	Interpatch	Patch	Interpatch
	7	0.5	
2.5			1.8
	4	1	
3			3
	2	1.5	
0.8			4
	6	1.6	
1			1.5
	1	1.7	
1.3			1
	3	1.2	
1			0.5
	5		
3.5			
	2		
1			
	0.5		
1			
	4		
1.5			
	15		
3			
	1		
4.5			
	9		
2			
	1		
2			
	3		
1			
	9		
2			

Transect No:3
Rangeland: CF
LOI: 0.581

Patch	Interpatch
	1.5
0.5	
	1
0.5	
	1
0.5	
	1
0.5	
	1
0.25	
	3
0.9	
	0.5
5	
	0.5
1	
	0.5
1	
	1
6.1	
	3
7	
	4
1	
	1.1
1	
	2
4	

Transect No:4
Rangeland: CF
LOI: 0.189

Patch	Interpatch
	4
1	
	2
0.25	
	6
1	
	5
0.5	
	1
0.25	
	3
0.25	
	2
0.5	
	2
2	
	2
0.5	
	1
1	
	3

Transect No:5
Rangeland: CF
LOI: 0.395

Patch	Interpatch
6.8	
	7
1.8	
	3
0.5	
	0.5
2.4	
	0.5
1	
	11
1	
	2.5
2.5	

Transect No:6 Rangeland: C LOI: 0.622	
Patch	Interpatch
2.4	
	5
3	
	2
3	
	0.5
4	
	4
3	
	1
2	
	0.5
4	

Transect No:7 Rangeland: CF LOI: 0.781	
Patch	Interpatch
13	
	3.5
7	
	0.5
1	
	1
1	
	0.5
5	
	2
1.5	
	2
12	
	1.8
7	
	0.4
13	
	0.3
1.7	
	1.6
4	
	7
0.6	
	1.2
12	
	2
2	
	0.3
5	

Transect No:8 Rangeland: CF LOI: 0.410	
Patch	Interpatch
2	
	7
1	
	0.5
2	
	9
3	
	4
2	
	3
4	
	2
0.6	
	1
1.2	
	0.5
3	

Transect No:9 Rangeland: CF LOI: 0.692	
Patch	Interpatch
42	
	1
3	
	14
7	
	4
1.2	
	2
5	
	3
3	
	5
4	

Transect No:10 Rangeland: CF LOI:0.578	
Patch	Interpatch
	6.4
2.5	
	0.5
1	
	0.8
3	
	0.7
16.8	
	2
7.5	
	0.8
5.6	
	1
21	
	7
3.5	
	26
1	

Transect No:11 Rangeland: CF LOI:0.661	
Patch	Interpatch
	2.5
3.4	
	0.5
0.8	
	6
0.5	
	1
10	
	0.5
7	
	3
1.5	
	4
2	
	2
3.1	
	5
3	
	7
11	
	2
23	

Transect No:12 Rangeland: CF LOI:0.661	
Patch	Interpatch
	2
3.5	
	12
6	
	1
23.5	
	12
1.5	
	0.8
7.5	
	4.5
13.5	
	6
9.5	
	3.1
2.2	
	7
22	
	6
9	

Transect No: 1 Rangeland: PC LOI: 0.323	
Patch	Interpatch
3	
	2
3	
	19
2.5	
	2
2.5	

Transect No: 2 Rangeland: PC LOI: 0.313	
Patch	Interpatch
	1
2	
	5
2.2	
	7
2.1	
	10
2	
	3
1	
	3
5	
	6
2	
	0.5
1.2	
	3

Transect No: 3 Rangeland: PC LOI: 0.565	
Patch	Interpatch
4	
	10.5
3	
	2
2	
	1.5
0.5	
	1
3	
	0.5
7	
	0.5
4	
	5
13	
	8
3	
	1
6	
	5

Transect No: 4 Rangeland: PC LOI: 0.483	
Patch	Interpatch
	4
3	
	5
3	
	8.5
3	
	2
4.2	
	1
2	
	1.2
4.1	
	5
3	
	3
5	
	4
4.2	

Transect No: 5
Rangeland: PC
LOI: 0.567

Patch	Interpatch
5	
	6.2
6	
	0.5
4	
	0.5
2.2	
	11
0.5	
	1
0.8	
	1
1	
	0.5
3	
	0.5
6	
	2
2	
	3
4	
	3
2	
	1.8
7	
	4
5	
	2

Transect No: 6
Rangeland: PC
LOI: 0.629

Patch	Interpatch/BS
	3
2.5	
	10
6	
	1
23	
	9
0.5	
	3
6	
	4.5
13	
	6
7	
	3.1
2.2	
	7
19	
	6
10	

Transect No: 7
Rangeland: PC
LOI: 0.556

Patch	Interpatch	Patch	Interpatch
3		4	
	1		7
4		0.8	
	1		8
3		4	
	13		4
17		2	
	14		0.5
16		4	
	20		
18			
	1		
10			
	1.2		
4			
	1.6		
2.5			
	1.8		
8			
	4		
5			
	5		
3			
	2		
1			
	10		
9			
	9		
6			

Transect No: 8
Rangeland: PC
LOI: 0.538

Patch	Interpatch
3	
	6
3	
	6
1.1	
	0.8
2	
	2
0.8	
	1
4	
	0.5
14	
	1.5
4	
	1
3	
	0.5
4	
	0.6
0.6	
	0.5
1.6	
	4
4.2	
	5
3	
	1
0.8	
	1.7
1	
	6
7	
	1
1.1	
	1
2.1	
	10
1	
	0.5
1	
	3

Transect No: 9
Rangeland: PC
LOI: 0.503

Patch	Interpatch
	0.3
2.1	
	1
1.4	
	5
7	
	9
3.1	
	2.8
4.2	
	2
6	
	3
4	
	5.5
4	
	4.1
8	
	6
0.6	
	3
5	
	2
6	
	3
2	
	6
5	

Transect No: 10
Rangeland: PC
LOI: 0.658

Patch	Interpatch
	2.2
2	
	0.5
0.6	
	4
0.6	
	1
10	
	0.5
7	
	2
1.5	
	6
2.2	
	3
3.1	
	4
3	
	8
11	
	2
23	

Transect No: 11
Rangeland: PC
LOI: 0.441

Patch	Interpatch
	8.5
20	
	3
4	
	11
3	
	5.5
1.2	
	1
6	
	22
3	
	10
14	
	4

Transect No: 12
Rangeland: PC
LOI: 0.536

Patch	Interpatch
	6
2	
	1
1	
	0.8
2	
	0.7
17	
	2
6	
	1
4	
	1
20	
	8
0.7	
	26
1	

Transect No:1
Rangeland: DC
LOI: 0.275

Patch	Interpatch
	3
0.9	
	2
0.7	
	1.2
0.3	
	2
1	
	8
1.1	
	1
1	
	0.5
0.5	
	0.5
1.3	
	1
1.2	
	1.9
0.5	
	1.9
0.25	
	0.5
0.25	
	0.5
0.25	
	1
0.25	

Transect No:2
Rangeland: DC
LOI: 0.219

Patch	Interpatch
	4
0.6	
	2
0.5	
	1
0.5	
	1
1	
	1
0.25	
	2
0.6	
	5
0.25	
	0.5
0.25	
	0.5
0.25	
	3
1	
	4
1	
	0.5
0.9	
	5
0.9	
	2
2.5	
	4

Transect No:3
Rangeland: DC
LOI: 0.243

Patch	Interpatch
	8
1.8	
	9
1.6	
	1.5
2.25	
	1
1	
	0.25
1	
	2
1.4	
	2.5
1.1	
	0.5
1	
	10

Transect No:4
Rangeland: DC
LOI: 0.282

Patch	Interpatch	Patch	Interpatch
	9		5
0.5		1	
	6		2
0.7		4	
	1.5		1
1.4		1.2	
	6		9
1.1		0.9	
	6		4
1		3	
	0.5		0.5
0.5		0.7	
	1		6
0.6		2	
	11		4
0.6		2	
	4		2
0.5		0.8	
	0.5		0.5
2		0.4	
	4		5
3		0.1	
	11		3
2		1.4	
	3		0.4
2		1.3	
	15		2
0.5		7	
	5		1
1		2	
	2		0.4
4		1.4	
	1		1
1.2		1.5	
	9		1
0.9		1.5	
	4		2
3		1.8	
	0.5		1
0.7		0.8	
	6		1
2		2	
	4		4
2		2	
	2		6
0.8			
	0.5		
0.4			
	5		
0.1			

Transect No:5
Rangeland: DC
LOI: 0.178

Patch	Interpatch	Patch	Interpatch
	33		4
2		0.5	
	4		2
0.5		0.4	
	0.5		14
0.6		3	
	14		13
0.6		0.8	
	2		0.5
0.7		19	
	2.5		
0.3			
	4		
0.5			
	19		
3			
	3		
1			
	17		
0.3			
	4		
5			

Transect No:6
Rangeland: DC
LOI: 0.241

Patch	Interpatch	Patch	Interpatch
	33		2
2		0.4	
	4		14
0.5		3	
	0.5		13
0.6		0.8	
	14		0.5
0.6		19	
	2		3
0.7		1.1	
	2.5		1
0.3		0.8	
	4		0.5
0.5		0.4	
	19		2
3		3	
	3		3
1		1	
	17		10
0.3		1.2	
	4		0.5
5		4	
	4		
0.5			

Transect No: 7
Rangeland: DC
LOI: 0.261

Patch	Interpatch
	2
3	
	4.5
0.5	
	1
2.5	
	12
3	
	28
2.2	
	12
6	
	0.5
4	

Transect No: 8
Rangeland: DC
LOI: 0.135

Patch	Interpatch
	5
3	
	11
0.5	
	7
4.5	
	1
0.2	
	8
0.2	
	21
1	
	0.5
0.5	
	1
0.2	
	1
0.2	
	8
0.8	
	2
0.8	

Transect No:9
Rangeland: DC
LOI: 0.450

Patch	Interpatch	Patch	Interpatch
	2		0.5
3		3	
	0.5		7
3		4	
	1		0.5
1		1	
	0.5		3
1.5		1	
	1.5		7
0.5		1	
	2		3
3.4		2	
	5		
2.5			
	0.5		
1			
	1		
1			
	3		
1			
	1		
2			

Transect No:10 Rangeland: DC LOI: 0.205			
Patch	Interpatch	Patch	Interpatch
	5	0.2	
3			2
	16	3	
0.2			3
	7	1	
5			16
	3	1	
1.1			27
	1	4	
1			4
	2	4	
0.5			21
	6	5	
1			5
	15		
2			
	12		
2.3			
	2		
1			
	4		
2			
	3		
4			
	2		
1.1			
	1		
2			
	15		

Transect No: 11 Rangeland: DC LOI: 0.313	
Patch	Interpatch
	5
4	
	7
5	
	6
2	
	12
4	
	16
6	

Transect No: 12 Rangeland: DC LOI: 0.419	
Patch	Interpatch
	4
2	
	14
8	
	13
1.1	
	1
4	
	12
3	
	3
4	
	2
3	
	0.2
8	
	5
6	

Where CF: former commercial farms; PC: pristine communal rangeland; DC: degraded communal rangeland