MAPPING POTENTIAL SOIL EROSION USING RUSLE, REMOTE SENSING, AND GIS: THE CASE STUDY OF WEENEN GAME RESERVE, KWAZULU-NATAL

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Declaration

I, Solomon G. Tesfamichael, wish to certify that the works reported in this thesis represent my original work except where acknowledged.

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Abstract

Accelerated soil erosion is drawing a growing attention with the recognition that the rate of soil loss is too great to be met by soil formation rate. Weenen Game Reserve (WGR) is an area with an unfortunate history of prolonged soil erosion due to excessive overgrazing that led to severe land degradation with prominent visible scars. This problem triggered the general objective of estimating and mapping potential soil erosion in WGR. Assessing soil loss in the area objectively has important implications for the overall management plans as it is reserved for ecological recovery.

The most important variables that affect soil erosion are considered as inputs in soil loss estimation models. In this study the RUSLE model, which uses rainfall, soil, topography, and cover management data, was employed. From the rainfall data, an erosivity factor was generated by using a regression equation developed by relating EI_{30} index and total monthly rainfall. The soil erodibility factor was calculated using the soil erodibility nomograph equation after generating the relevant data from laboratory analysis of soil samples gathered from the study area. Using exponential ordinary kriging, the point values of this factor were interpolated to fill in the non-sampled areas. The topographic effect, which is expressed as the combined impact of slope length and slope steepness, was extracted from the DEM of the study area using the flow accumulation method. For mapping of the land cover factor, *in situ* measurements of cover from selected sites were undertaken and assigned values from the USLE table before being related with MSAVI of Landsat 7 ETM+ image. These values were then multiplied to get the final annual soil loss map.

The resulting potential soil loss values vary between 0 and 346 ton ha^{-1} $year^{-1}$ with an average of 5 ton ha^{-1} $year^{-1}$. About 58% of the study area experiences less than 1 ton ha^{-1} $year^{-1}$ indicating the influence of the highest values on the average value. High soil erosion rates are concentrated in the central part extending as far as the south and the north tips along the eastern escarpments and these areas are the ones with the steepest slopes.

The results indicate a high variation of soil loss within the study area. Nevertheless, the majority of the area falling below the average might foresee that the soil erosion problem of the area can be minimized significantly depending largely on soil management. The most important areas for intervention are the medium and low erosion susceptible parts of WGR, which are mainly found in the flatter or gently sloping landscapes. The steepest areas are mostly covered with rocks and/or vegetation and hence less effort must be spent in managing them. Overall, the reported increasing density of the vegetation community in the area that reduces the exposure of soil from the impact of direct raindrops and surface-flowing water must be pursued further.

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ACRU	Agricultural Catchments Research Unit
AGNPS	AGricultural Non Point Source pollution model
amsl	above mean sea level
ANSWERS	Areal Non-point Source Watershed Response Simulation
BEEH	Bioresources Engineering and Environmental Hydrology
BF	Burst Factor
BRG	BioResource Group
CREAMS	Chemicals, Runoff, and Erosion from Agricultural Management Systems
DEM	Digital Elevation Model
DTM	Digital Terrain Model
EPIC	Erosion Productivity Impact Calculator
ESRI	Environmental Systems Research Institute
ETM	Enhanced Thematic Mapper
EUROSEM	EUROpean Soil Erosion Model
FAO	Food and Agricultural Organization
GIS	Geographical Information System
GLASOD	GLobal Assessment of SOil Degradation
GUEST	Griffith University Erosion System Template
HEM	Hillslope Erosion Model
IDW	Inverse Distance Weighting
KZNNCS	KwaZulu-Natal Nature Conservation Service
MOMS	Modular Optoelectronic Multispectral Stereo-scanner
MSAVI	Modified Soil Adjusted Vegetation Index
MUSLE	Modified Universal Soil Loss Equation
NDVI	Normalized Difference Vegetation Index
RUSLE	Revised Universal Soil Loss Equation
SAVI	Soil Adjusted Vegetation Index
SEMMED	Soil Erosion Model for MEDiterranean areas
SLEMSA	Soil Loss Estimation Model for Southern Africa

TM	Thematic Mapper
TNDVI	Transformed Normalized Difference Vegetation Index
TSAVI	Transformed Soil Adjusted Vegetation Index
UGR	Umfolozi Game Reserve
UK	United Kingdom
UNEP	United Nations Environmental Program
UNESCO	United Nations Educational, Scientific, and Cultural Organization
USA	United States of America
USLE	Universal Soil Loss Equation
WEPP	Water Erosion Prediction Project
WGR	Weenen Game Reserve
WGS	World Geographic System

CHAPTER ONE: INTRODUCTION

1.1 OVERVIEW OF SOIL EROSION

Soil erosion is the removal of soil by water and wind (Zachar, 1982; Lal, 1994a). The erosive action includes two processes: detachment and transport of soil particles (Morgan, 1995; Sharma, 1996). Blum (1998) and Morgan (1995) define soil degradation that includes, primarily, soil erosion as the loss of soil's capability to give sustained productivity or utility, which can be induced naturally or caused/accelerated by anthropogenic activities.

Erosion is not a recent phenomenon; it is rather in close relation with mankind (Troeh *et al*, 1980; Morgan, 1995; Blum 1998) and worsening with the increase in world population (Hudson, 1995; Lal, 1998a). Man's aggravation of soil erosion "directly" (accelerated) and "indirectly" (natural but triggered by man) is mostly a result of agricultural needs and urban developments that resulted in the modification of the earth's surface (Golomb and Eder, 1971). Because of the mismatch between natural resources and population growth (Jacks 1986; Lal, 1994a), the exploitation of existing farmland with the extent that exceeds its capacity is inviting man to encroach on lands previously considered as marginal (Troeh *et al*, 1980). The impact of such soil loss on the living condition (not to mention the associated environmental deterioration) of human being is enormous and is felt on the site where the soil is lost and at destination sites (Morgan, 1995).

1.2 SOIL EROSION ASSESSMENT

Parallel to the awareness of the soil erosion problem people have long been trying to sustain agricultural productivity while fighting soil loss (Higgitt, 1992). In modern days, fighting soil loss has based itself on the result of an appraisal of the problem (Higgitt, 1991). Clearly, this approach helps where and how to act depending on the nature and extent of the problem. Assessment of soil loss has grown from the simplest and qualitative methods to more comprehensive and quantitative methods. Recently, the vision in soil erosion assessment has turned towards accounting for much more complex parameters, which, no doubt, provide a more reliable estimation, with the need for large input data (Lorentz and Schulze, 1994).

Of all the approaches, the USLE and its successor the RUSLE, are the most widely applied in the world (Lorentz and Schulze, 1994; Lorentz and Howe, 1995; Smith, 1999; Ritchie, 2000) despite their development from the database of the USA. These models were developed to help guide conservationists in the design to fight soil loss (Wischmeier and Smith, 1978; Renard *et al*, 1997). Though the USLE was earlier considered unsuitable for Southern Africa by Elwell (1978), Smith (1999), in his revision of empirical models, concluded the RUSLE model to be potentially robust if applied in South Africa.

While soil erosion assessment is yet a difficult task (Higgitt, 1991) because of the complexity of the factors governing the problem, it is becoming an interesting field of study credited to its integration with remote sensing and Geographic Information Systems (GIS) technologies (Rooseboom *et al*, 1992).

The use of GIS appears to be a necessity in soil erosion assessments (De Roo *et al*, 1989; Flacke *et al*, 1990) as the latter involve spatial variation due to nature's complexity and management impact. In most recent times nearly all soil erosion assessment methods are integrated well with GIS because of the simplicity of spatial analyses.

Remote sensing has been used in land degradation studies by using ground based and aerial photos (Ritchie, 2000). The technique makes use of sensed spectral difference of ground features to interpret and classify an area based on its erosion status. Ritchie (2000, pp. 276-277) summarized the application of the technology in soil erosion assessment to be in the form of "photointerpretation / photogrammetry, model/GIS inputs, spectral properties, and topographic measurements." With time, this technology increased its involvement by providing more reliable information that can be quantified with better confidence. The technology is enjoying its wide acceptance for many reasons including its coverage of large and inaccessible areas, collection of data within short periods of time that helps take repeated information from a given area within a year, collection of data in a continuous form which leaves no space for non-measurement-points, and provision of data that can readily be read and analyzed in a GIS environment (Engman, 1995).

1.3 SIGNIFICANCE OF SOIL EROSION ASSESSMENT AND THE STUDY

Recognizing the existence and the danger of soil erosion and its effects is only the first step towards any intervention intended to tackle the problem (Oldeman, 1994). Assessing soil erosion helps identify areas and extent of erosion hazard (Scotney, 1978). Such assessment does not try to give the exact picture of soil erosion; instead it provides a reasonable estimation (Morgan, 1995). The significance of assessment models for evaluating the severity of erosion makes them of great importance since they could be used for selection of conservation practices (Troeh *et al*, 1980; Smithen, 1981; Morgan, 1995; Russell *et al*, 1995).

In South Africa several researchers have carried out soil erosion assessments. It is acceptable that most of such works are undertaken in areas where there is reasonably adequate data for doing so or on experimental plots with complete control over the variables. Weenen Game Reserve might be one of the areas that are short of the information needed and hence, except for limited occasional and qualitative ones, no soil erosion assessment has ever been reported. Also, most similar studies carried out elsewhere are at small scale such as at river basin, regional or national level.

This study entirely sets its objective to make the first quantitative soil erosion assessment using the available data in WGR as opposed to few, non-objective, and qualitative assessments that relied on subjective judgments. The relatively small size of the area has also permitted a detailed study. It is also anticipated that baseline information that might be made available by this study could be used in current and future projects in the study area. Apart from these, the results of this study may have major contributions towards the long-term management strategy of the area. Furthermore, some of the approaches followed in this study can be applied in other parts of South Africa.

1.4 SOIL EROSION AT WEENEN GAME RESERVE (WGR)

Weenen Game Reserve (WGR) had been under extensive use since 1850 until it was taken over by the Department of Agriculture in 1948 (Camp, 2001). The major activity that degraded the area in a century was overgrazing (Weenen Game Reserve: Visitors' Guide, 2003). The area served as a "labor farm" given to tenants by landowners (Camp, 1995b; Camp, 2001). These tenants together with their livestock, in addition to the ones owned by the landowners, used to stay totally dependent on the area in exchange for their services during six months per year of no-farm labor season. To make matters worse the number of farmers together with their livestock size was increasing and this was welcome by the landowners who needed more labor. In 1937, there were 177 families with 1000 people. These people owned more than 2460 animal units (Camp, 2001).

Realizing the severity of the degradation, the area was taken over by the Department of Agriculture in 1948 after which the area was used for research purposes to help demonstrate soil recovery. Several soil reclamation techniques were undertaken over the next 25 years with tremendous effects on reducing soil erosion and gradual recovery of the natural veld (Camp, 2001; Weenen Game Reserve: Visitors' Guide, 2003). In 1975 the local community and the Weenen Town Board decided to put the area under the administration of the KwaZulu-Natal Nature Conservation Service (KZNNCS). The KZNNCS has fenced the reserve area with a 1.8 m veldspan (Weenen Game Reserve: Visitors' Guide, 2003).

With all the appreciable successes, it is equally reported and witnessed that the reserve is yet experiencing or highly exposed to soil erosion. This implies that there shouldn't be any room for complacency due to the achievements made so far. Instead, regaining the old natural wealth of the area should be envisaged, which can be materialized through a concerted effort and sound management. To this end, base line studies that revolve around the natural resources of the game reserve have positive contribution in management strategies. Likewise, the results of this study are expected to give important indicators as to how and where soil conservation should be prioritized.

1.5 AIM AND OBJECTIVES

The aim of this study is to assess in detail soil erosion at WGR using remote sensing, GIS and the RUSLE model and provide background information for decision-making in various environmental related management activities. The specific objectives of this study are:

1. To gather relevant data to be used in quantifying the factors of the RUSLE model.
2. To quantify the most important soil erosion factors in WGR.
   - To develop a rainfall erosivity equation specific to the area.
   - To calculate the soil erodibility values of selected soil samples.
   - To quantify the effect of topography on soil erosion.
   - To derive an equation to map cover factor.
3. To produce maps showing the distribution of the soil erosion factors.
4. To combine the soil erosion factors based on the RUSLE equation and estimate the annual soil loss rates.
5. To show the areal distribution of soil loss in WGR and classify the area according to the intensity of soil erosion reflecting the variation in erosion status across the area.
6. To prioritize areas with high erosion hazard so that such areas get faster management intervention. This is one reason that the RUSLE model is applied. Once areas of high erosion risk are identified then they can be managed to prevent further deterioration.

1.6 STRUCTURE OF THE THESIS

This study is presented in six chapters. Chapter two discusses soil erosion related issues. Though the main purpose of this study is soil erosion assessment, the literature review includes more information on the subject. Thus, types, driving forces, causes and effects of soil erosion are first reviewed to show the extent of the problem in different places. Following this, various soil erosion assessment methods are discussed. Finally, a number of soil erosion assessments that have strong relevance to this study and undertaken in different parts of the world are outlined.

In Chapter three, the study area based on available information is described. The description includes the geographical location, the settlement history, climatic conditions, the physical set-up, and biotic composition of the study area.

Chapter four provides a detailed description of the techniques employed in this study. The chapter describes the data sets used and the procedures that ultimately led to the final results.

In Chapter five the findings of the study are reported and are followed by discussions. The results are mostly displayed in the form of maps wherever possible whereas various works are brought into the discussion part to compare the results.

Chapter six comprises two parts: conclusions and recommendations. The conclusions part provides deductions based on the results whereas in the recommendations part management intervention areas are pointed based on the severity and the reversibility of the problems.

CHAPTER TWO: LITERATURE REVIEW

Soil erosion is recognized as a serious environmental problem that is costing the livelihood of people in different ways. It has a universal nature that scholars from different corners of the world are obliged to write about. While environmental consequences of the problem are relatively new concern, agricultural productivity losses have long been the main focus. In addressing the problem researchers use appraisal methods that vary from assessing the actual soil loss to studying the effects on different related resources. This chapter deals with the revision of the related literature on the problem. Though the main goal of this revision is on assessment of soil loss, it also includes the basic knowledge on the problem. This is followed by discussion of the most widely used assessment models and the various case studies that employed these models.

2.1 SOIL EROSION AND PERIODS OF THE PROCESS

Based on the period of the process, two broad divisions are given to describe the danger of soil erosion. These are natural/geologic and accelerated/human induced (Zachar, 1982; Hudson, 1995; Morgan, 1995; Blum, 1998).

2.1.1 Geologic soil erosion

Natural rate of erosion is given the name because its rate is in dynamic equilibrium with soil formation rate (Murgatroyd, 1979; Beckedahl *et al*, 1988). This is a universal phenomenon that has given the earth its present shape (Wild, 1993), and is also taken as an important and the first step in the soil formation process (Troeh *et al*, 1980; Schwab *et al*, 1993), because the occurrence of weathering responsible for breaking rock material down to form new soil needs direct contact of climatic factors and the rock. Because of the long period of time involved for this type of soil erosion, it is usually difficult to notice the consequences and hence little/no effort is made to stop such soil loss. The average annual soil loss by geologic process is usually considered to be less than 1 m ton ha^{-1} $year^{-1}$ (Troeh *et al*, 1980). South Africa's estimate of natural soil loss is about 0.5 to 1.4 cm per 1000 years (Braune and Looser, 1989, cited in Scotney and McPhee (1992)) whereas Murgatroyd (1979) estimated soil loss of 0.16 ton ha^{-1} $year^{-1}$ for the Tugela basin, KwaZulu-Natal.

2.1.2 Accelerated soil erosion

Accelerated erosion, which by far exceeds the soil formation rate (Thornes, 1985), is posing serious threats to deplete the entire natural resource base in all parts of the world with alarming reported estimations (Lal, 1994a). This type of erosion is a direct result of human activities for the quest of better life or survival (Wild, 1993; Hudson, 1995) and hence it is also called human induced erosion (Zachar, 1982). Maud (1978) suggested that the past 200 years of human dependence on land had significant effects resulting in the present degraded soil conditions in KwaZulu-Natal. Soil that would normally be washed or blown away in many years is being lost in less than a year, exceeding soil formation rates by far (Jacks, 1986). To emphasize the rate of accelerated erosion as caused mainly by deforestation and agriculture, Piemetel (1976, cited in Goudie (1981)) estimated 30 ton ha^{-1} $year^{-1}$ soil losses in the USA, which were approximated to be 8 times greater than soil formation. In South Africa accelerated soil loss, which is 4 times higher than the world's average, is estimated to be 12 cm per 1000 years, and this is about 20 times the rate of soil formation (Braune and Looser, 1989, cited in Scotney and McPhee (1992)).

2.2 TYPES OF SOIL EROSION

2.2.1 Inter-rill erosion

Nearing *et al* (1994) and Troeh *et al* (1980) define inter-rill or sheet erosion as detachment of soil by raindrops to be moved by shallow sheet flows. The first phase of inter-rill erosion is detachment of soil by splash erosion (Zachar, 1982; Sharma, 1996). Factors that favor the displacement of soil after detachment are topographic, namely, steepness and length of slope (Gerits *et al*, 1990; Torri, 1996), runoff rate, and surface roughness (Nearing *et al*, 1994). Gerits *et al* (1990) reported that as slope angle increases there is increased portion of rainfall resulting as runoff having high velocity because of the steepness and flow depth.

Such erosion occurs where there is high rainfall amount that exceeds infiltration capacity and surface depression storage (Morgan, 1995). Depending on this, a land with no pronounced channels may have runoff with smooth depth of water (Zachar, 1982; Morgan, 1995).

Consequently, the depth of the removal of soil from the whole area becomes uniform provided that the other soil properties are the same (Troeh *et al*, 1980; Schwab *et al*, 1993; Laflen and Roose, 1998).

Meyer (1985) reported that inter-rill erosion decreases with continued rainfall. This reduction was attributed to the fact that easily erodible soils are eroded whereas the more resistant ones remain uneroded. According to Morgan (1995), generally, overland flow occurs where the area is rich in finer materials. Yet, it is important to note that the detachment capacity of inter-rill flow is smaller than detachment by raindrops because of lower shear force in the former case (Sharma, 1996). Since the water depth is shallow the surface roughness causes the flow to vary drastically within a very short distances resulting in considerable scours (Nearing *et al*, 1994; Morgan, 1995).

Most overland flow is related to semi-arid areas or areas with sparse vegetation cover (Nicolau and Asensio, 2000). Thus, vegetation removal becomes responsible for the increase in overland flow (Morgan, 1995). Nearing *et al* (1994) mentioned the occurrence of such erosion on rangelands and no-till situations or on gentle and short-lengthed slopes. According to Sharma (1996) reducing rainfall energy by 89%, without decreasing rainfall amount, inter-rill soil loss decreased by 90% or more.

2.2.2 Rill erosion

Rills, which usually range between 100 to 300 mm wide and 50 to 150 mm deep (Foster *et al*, 1985), result as overland flow breaks up and forms channels (Morgan, 1995) and can usually be overcome by conventional cultivations (Schwab *et al*, 1993; Wild, 1993; Grissinger, 1996). Regarding its formation, however, Morgan (1995) also reported a case in mid-Bedfordshire where rills developed near the bottom of a slope and extended upslope. Sticking to the first notion of rill formation, subsequent development and convergence of flow paths intensify particle movement that causes scouring and channeling. As the channel depth gets deeper, the flow concentration increases to detach more soil particles (Wild, 1993). Morgan (1995) again echoed studies that showed stages of rilling: unconcentrated overland flow, concentrated overland flow paths, micro-channels without head cuts, and micro-channels with head cuts. Since rills, whose sediment source is mainly from inter-rill areas, have concentrated flows, they can transport large grains minimizing size selectiveness (Lal and Elliot, 1994; Morgan, 1995). Morgan (1995) cites the findings of Meyer *et al* (1975) that reported that 15% of particles carried in rills of 3.5° slope were larger than 1 mm size and that 3% of the particles were larger than 5 mm.

In general, rills account for the most part of erosion (Morgan, 1995). In the USA a study on a 4.5 m long plot indicated that 80% of sediment was moved in the rills. Schwab *et al* (1993) point that rill erosion is predominant in cases of intense storms occurring on soils of high-runoff-producing characteristics and easily erodible topsoil. The sediments enter into rills partly from inter-rill areas by overland flow and rain splash (Morgan, 1995). The same author puts the works of Foster and Meyer (1975) to show that up to 87% sediment in rills may be derived in this way. Whereas, in Belgium it is estimated that half of the material removed from inter-rill areas enter the nearby rills (Morgan, 1995).

The significance of rills is related to their intensity in a given area (Morgan, 1995). In Presov, former Czechoslovakia, rills that were spaced 0.330 m apart accounted for 58% of soil erosion that damaged 91 ha of land. Again in Banska Bystica, of the same country rills contributed 70% of the total sediment that affected 60% of a potato field. In contrast rills spaced 300 to 350 m intervals in mid-Bedfordshire, England accounted for only 20 to 50% of sediment loss.

2.2.3 Gully erosion

Gullies are steep-sided channels subject to intermittent flash floods (Hudson, 1995). Unlike rills, gullies are relatively permanent and difficult to reclaim (Zachar, 1982; Wild, 1993; Hudson, 1995; Morgan, 1995).

Formation of gullies is so complex to exactly define its process (Zachar, 1982; Goldman *et al*, 1986; Morgan, 1995; Oostwoud Wijdenes and Bryan, 2001). It usually starts with small depressions or knicks that allow water concentrations. Concentrated water then joins several depressions and forms channels. Most erosion, here, occurs at the base of the scarp resulting in deepening of the channels, which, in turn, leads to collapse of the headwalls upslope. Further down, sediment loss continues by the scouring action of running water and its sediment, and by falling banks attributed to water saturation. Gullies also form due to sub-surface actions that create tunnels and pipes (Wild, 1993). Morgan (1995) reported that a study in Hong Kong showed that most water was removed from hillsides by sub-surface flow, and heavy rain that fell afterwards resulted in subsidence of the surface leaving the pipes as gullies. Such pipe or tunnel to be a cause for gullying is also observed in the Sudan, the USA, and Hungary (Morgan, 1995). Gully formation by sub-surface flow is common on slopes where the subsoil has low permeability as in the case of high clay content in the B horizon (Wild, 1993).

Landslides are other causes of gully formations. In this case the resulting landform may serve as a channel for running water. Such gully formation was studied in Italy and in Sweden (Morgan, 1995).

The main driving force in gully formation is excessive water on a given land surface that may be caused either by changing rainfall amount or vegetation clearance (Morgan, 1995). Change in rainfall implies higher or lower rainfall to limit vegetation growth, whereas vegetation clearance is usually related to deforestation, overgrazing, and burning of vegetation (Morgan, 1995). Boardman *et al* (2003) reported that overgrazing during the past in the Sneeuberg uplands of Great Karoo, South Africa, is responsible for major gully formations.

The impact of gullies is serious as studied in China by Dicenzo and Luk (1999) who found that gullies accounted for 85% of the sediment load that passed through a small dam in the area. In the same way Martinez-Casasnovas *et al* (2002) reported that gully erosion, which covered only 15% of their total study area, resulted in 50% of the total soil loss as compared to 42% loss that affected 39% of the total area in the Alt Penedes-Anoia region, Spain. Watson and Ramokgopa (1997), after identifying 10 gully erosion-stricken and 7 other susceptible areas in the Mfolozi catchment of KwaZulu-Natal, South Africa, recommended that these areas should not be allocated under the Land Reform Programme. Additional 15 subcatchments were also found to have land types exposed to gully formations.

2.2.4 Landslides and mass movements

Landslides occur on areas where the soil is very loose to be affected easily by gravitational force implying that steepness is an important factor. A study carried out in northern Ethiopia gave reasons such as lack of lateral support lost due to gullying, presence of impervious geological layers, and laterally penetrating water as the causes of mass movements (Nyssen *et al*, 2003). Mass movement erosion has so far little attention and documentation but has a significant effect on land quality as studied by Blaschke *et al* (2000). Their finding in New Zealand show that up to 80% and up to 20% production loss at field scale and at farm scale, respectively on pastoral grazing can be caused by mass movement whereas in the Pacific Northwest coastal forests of North America such erosion has resulted in the decline of forest wood production by 35-50%.

2.3 FACTORS THAT INFLUENCE SOIL EROSION

The immediate factors that influence soil erosion are rainfall erosivity, soil erodibility, topography, plant cover, and conservation practice (Morgan, 1995). These factors are examined in the following sub-sections.

2.3.1 Erosivity

Erosivity is the ability of eroding agent to detach and transport soil (Lal and Elliot, 1994; Morgan, 1995). Rainfall and wind are the two important eroding agents of which water erosion is the dominant one (Hudson, 1995). The contribution of rainfall to soil erosion has two phases: detachment and transportation commenced in their respective order (Lal and Elliot, 1994). Detachment of soil particles occurs as the raindrop strikes the soil. Hence, this part of erosion depends on the energy of the raindrop that, in turn, is the direct result of the speed and the drop size (Zachar, 1982; Lal, 1994a). Elaborating, the action of rainfall erosivity, Troeh *et al* (1980) gave three effects. First, a raindrop breaks aggregates into smaller aggregates and/or individual particles. Second, it displaces the detached particles as water splashes back into the air. Third, a raindrop compacts the soil aggregates to create surface crust thereby reducing infiltration capacity.

The second step of erosion due to rainfall is transportation. Once soil particles are detached they are liable to be displaced. Water excess from infiltration and water holding capacity of the soil results in runoff (Onchev, 1985; Morin, 1996). A steep land with minimum vegetation cover or surface roughness allows the surplus water to flow down the slope carrying with it the detached soil particles.

Nevertheless, it is difficult to specifically tell the important characteristics of rainfall erosivity. Ruangpanit's (1985) work at Doi Pui, Kog-Ma Watershed, northern Thailand recognized the existence of the direct relationship between amount, duration, and intensity of rainfall on the one hand and runoff and erosion on the other. Yet, in most cases, it is the rainfall intensity rather than the amount that is directly linked with soil loss (Goldman *et al*, 1986; Morgan, 1995). According to Goldman *et al* (1986), intense rainfall over a short period of time is more erosive than high rainfall amount over long-duration of time with low intensity. Earlier, based on an experiment in Tien Shuy it is stated in Zachar (1982) that with an increase in rainfall intensity, surface runoff grew and led to increased soil erosion.

Morgan (1995) reported that rain falling on an area, which is already saturated with previous moisture, results in runoff regardless of its amount or intensity. The next question is finding the threshold value of rainfall erosivity. Studies in various places showed that 25 mm hr^{-1} is the critical value, which is a high value for England and Germany that use 10 mm hr^{-1} and 6 mm hr^{-1}, respectively. In developing a universal index of erosivity, Onchev (1985) specifies a rainfall intensity of 10.8 mm hr^{-1} and a storm depth of greater than or equal to 9.5 mm as erosive and should be used to derive the rainfall erosivity factor. This approach was shown to have the highest correlation (r=0.877) of all indices including the Universal Soil Loss Equation's EI_{30} index.

2.3.2 Erodibility

Erodibility is the resistance of a soil to detachment and transport (Zachar, 1982). Le Bissonnais (1996) and Lal and Elliot (1994) define soil erodibility as the measure of soil susceptibility to detachment and transportation. Although soil erosion depends on various factors, erodibility, which refers to the inherent properties of the soil, is considered one of the most important ones

(Le Bissonnais, 1996). The properties included in erodibility are texture, structure, shear strength, infiltration capacity, organic matter, and chemical content (Troeh *et al*, 1980; Goldman *et al*, 1986; Lal and Elliot, 1994; Miller and Gardiner, 1998). The same sources define soil texture as the relative size and distribution of sand, silt, and clay. Large particles such as sand are erosion resistant because of the force required to displace them, and fine particles such as clay resist erosion because of the difficulty to detach them due to their cohesiveness (Lal and Elliot, 1994). But, once clays are detached from soil aggregates, they are easily transported in suspension until they reach stagnant water (Goldman *et al*, 1986). Silts, which are classified in between sands and clays, are the most erodible soil particles (Troeh *et al*, 1980; Lal and Elliot, 1994). Soils with 40-60% silt are highly susceptible to erosion (Morgan, 1995). Researchers like Evans (1980, cited in Morgan (1995)) preferred to use clay content for determining soil erodibility and defined that soil with 9-30% of clay is taken as the most erodible. Defining erodibility based on clay content looks fairly acceptable since clays readily combine with organic matter content to form a stable aggregation that is significant to reduce detachment and thereby erodibility (Troeh *et al*, 1980; Morgan, 1995).

Another soil property that influences erodibility is shear strength, which is defined as a measure of soil's cohesiveness and resistance to shearing forces (Morgan, 1995). Forces that create shear strength are gravity, moving fluid, and mechanical loads. Whenever an external force is applied on soil, the particles tend to slide over one another or break the interlocking forces that include a frictional force (Morgan, 1995). This frictional force is the strength that varies for different soils and hence determines the overall shear strength. Shear strength is also influenced by plant roots and this was shown by Mamo and Bubenzer (2001) who found out a 20% greater strength for corn and soyabean plots than for fallow plots.

Because of their effects on aggregate stability, organic and chemical contents of the soil are important in determining infiltration capacity (Morgan, 1995; Le Bissonnais, 1996). Siegrist *et al* (1998) conducted an experiment to assess the effect of organic agriculture on soil erodibility and concluded that organic plots had better aggregate stability.

The other important factor affecting soil erodibility is the type of exchangeable cations that affect dispersion/flocculation (Le'Bissonnais, 1996; Norton *et al*, 1999). These authors report that Al^{3+} and Ca^{2+} are known for their binding effects on soil particles creating well-structured soil aggregates, whereas Na^+ is a dispersal ion that breaks soil aggregation.

2.3.3 Slope

Topographic factors of an area can be characterized by steepness, slope length, slope aspect, and slope shape (Zachar, 1982; Morgan, 1995; Torri, 1996). There is agreement that soil erosion increases directly with slope length and steepness. A steep slope implies a high runoff velocity, and a long slope results in increased runoff volume (Goldman *et al*, 1986). Zachar (1982) reported that doubling the steepness of the slope increases the kinetic energy of runoff proportionally but the carrying capacity increased 32 times.

In addition, raindrops splash soil particles more downslope than upslope and the extent is directly related to the steepness. Morgan (1995) reports that the top of the slope is usually considered as no erosion belt. At some distance from top slope sufficient water accumulates on the surface to start flowing, which gets deeper as the length of the slope increases (Morgan, 1995).

2.3.4 Plant cover

The role of vegetation in soil erosion prevention is in interception of the raindrops (Goudie, 1981; Stocking, 1994; Morgan, 1995), obstructing surface runoff (Goudie, 1981; Zachar, 1982), increasing infiltration (Morgan *et al*, 1997; Haigh and Sansom, 2000; Loch, 2000), and binding the soil particles by the root systems (Goudie, 1981; Stocking, 1994). Loch (2000) reported that vegetative cover reduced soil erosion in Australia from 30-35 ton ha^{-1} at 0% vegetative cover to 0.5 ton ha^{-1} at 47% cover. Similarly, Mamo and Bubenzer (2001) compared soil detachment of soyabean plots with fallow soils and found out that the former had one half detachment of the latter. The interception of raindrops breaks them down and reduces the kinetic energy before the raindrops reach the surface (Troeh *et al*, 1980; Stocking, 1994). To prove this an experiment was made at a research station in Zimbabwe, where mean annual soil loss from bare ground was 4.63 kg m^{-2} compared with 0.04 kg m^{-2} from *Digitaria* covered land within the same period of time (Morgan, 1995). The mosquito gauze experiment of Hudson and Jackson (1959, cited in Morgan

(1995)) was tested on two identical bare plots, but over one plot fine wire gauze was suspended. The results showed that the mean annual soil loss over a ten-year period for the open plot was 126.6 kg m^{-2} and 0.9 kg m^{-2} for the gauze-covered plot. Thus, the wire gauze reduced the impact of raindrops on the ground by absorbing the energy.

Vegetation contributes to soil loss prevention depending on its height, canopy cover, and root density (Morgan, 1995). Increasing the canopy alone may worsen erosion unless there is enough litter to intercept the drops (Troeh *et al*, 1980). A canopy as high as 7 m can maintain 90% of terminal velocity of raindrops (Morgan, 1995). In addition, raindrops intercepted by leaves usually form a larger drop that exerts greater force on a bare soil below than the drops that reach the leaves (Foster *et al*, 1985; Stocking, 1994; Morgan, 1995). A 3.1 times of open ground soil detachment was observed under beech canopy (Morgan, 1995). In addition, tall trees usually reduce ground cover by their shading effect on understorey growth and this leads to unobstructed surface runoff and exposure of the soil to raindrops (Stocking, 1994). Zhou *et al* (2002) from a study in southern China noted that *Eucalyptus* increased the kinetic energy of the raindrop that led to increased soil erosion when the rainfall amount was greater than 5 mm and rainfall intensities were less than 20 mm hr^{-1}. Such explanation might lead to preference of plants based on the height and size of canopy. For example, according to Morgan (1995) a study showed that cotton reduced the volume of rain reaching the ground but increased the median drop size. Foster *et al* (1985) reported that an increase in drop size even with short canopies might not be followed by increased soil erosion since raindrops falling from these canopies have low velocity.

The other effect of vegetation on soil erosion is acting as a barrier against surface runoff (Goudie, 1981; Morgan, 1995). Runoff is slowed down only if the obstruction is on the surface that depends on the surface vegetation cover (Troeh *et al*, 1980). Although runoff is not completely blocked, vegetation creates roughness that reduces runoff velocity significantly (Zachar, 1982). Moreover, vegetation due to its improvement of granulation increases infiltration rate that has an effect in reducing runoff (Stocking, 1994; Morgan, 1995). A well granuled soil has a higher porosity that improves moisture-holding capacity (Troeh *et al*, 1980; Zachar, 1982). Thus, it is clear that removing vegetation for any purpose results in increased surface runoff and consequently soil erosion (Goudie, 1981).

2.4 CAUSES OF SOIL EROSION

The main driving forces of soil erosion are wide range of human activities that tend to disturb natural stabilities. The causes range from removal of vegetation cover and its impacts to shaping of surface topography that might induce or accelerate soil displacement and movement. The most important causes are summarized separately in this section.

2.4.1 Agriculture

From all natural components land is the first asset that a human being acts on. The most important reason behind this is agriculture (Miller and Gardiner, 1998). Cultivation in South Africa, for example, on erodible soils has been one of the main reasons for soil erosion (Scotney and McPhee, 1992; Laker, 2000). Agriculture requires considerable land and every new land has to be obtained by clearing a relatively well-established natural forest areas. For example, five sixth of South Africa's total land is under agricultural use (Adler, 1985). Agriculture can be linked to soil erosion in different ways some of which are explained as follows. First, removing the natural vegetation cover exposes the soil to any kind of external actions that lead to its removal (Troeh *et al*, 1980). Heat, rainfall, and wind readily make their actions and erode the soil.

The second effect of agriculture on soil erosion is cultivation through disturbance of the top layer of soil, which should be given more attention. Since crops are seasonal or short-lived, they are removed after the harvest is exploited, every time of which leaving the soil loose and easy for dislocation. So, two effects are registered here, namely, breaking up of the soil particles as compared to the well-stabilized characteristics under natural condition and exposing the ground surface unlike the full canopy and litter cover of the natural vegetation. To study the effect of cultivation on soil loss, Choudhary *et al* (1997) took three soil samples representing mouldboard-ploughed, chisel-ploughed, and no-till soils from a Wooster silt loam of Ohio, which was under corn cultivation for 32 years. After subjecting the samples to a simulated rainfall, soil erosion and runoff were found to be the highest for mouldboard-ploughed and the least for no-till soil.

The third effect can be related to the system of tillage. Tapia-Vargas *et al* (2001) examined the impacts of various combinations of residue cover and tillage levels on runoff and sediment yield in Mexico. The treatments were conventional tillage, no-till with zero residue cover, no-till with 33% residue cover, and no-till with 100% residue cover. The results showed that the first two treatments had higher runoff and sediment yield leading to the recommendation of no-till farming.

2.4.2 Overgrazing

In addition to cultivation, overgrazing is also an important cause of vegetation clearance and the subsequent soil erosion. This is the main cause in South Africa (Laker, 2000). Uncontrolled or poorly managed grazing brings about removal of vegetation that exposes the soil to all types and processes of erosion. Bari *et al* (1995) conducted an experiment in Pakistan by applying various levels of grazing on different plots and found that no-grazing treatment produced the least sediment accumulation. In southern Iceland, Simpson *et al* (2001) conducted a study to examine the effect of grazing on soil erosion and reported that the management aspect of grazing is more responsible than the stock numbers, a view shared by Zobisch (1993) from a study in Kenya. The work of McEldowney *et al* (2002) in Colorado, USA related high amount of sediment deposition with decrease in stem density, due to increased overland flow from grazed field as compared to the non-grazed.

Another major problem usually related to livestock is their trampling effect as assessed by Eldridge (1998). The researcher used a sheep's hoof at different levels to represent stocking rates and found that trampling increases soil erosion significantly. The results of this study indicated that trampling accounted for 33% of the total soil erosion. In assessing grazing effects on soil physical properties in eastern Nigeria, Asadu *et al* (1999) revealed that trampling compacted the soil surface under study and reduced the macroporosity and total porosity resulting in reduced hydraulic conductivity. Compared to non-grazed arable fields the reduction in total porosity and hydraulic conductivity amounted to 15 and 60%, respectively.

2.4.3 Fire

Johansen *et al* (2001) conducted a study in New Mexico, USA to examine the effects of fire on runoff and soil erosion. The findings revealed that 45% of the applied 120 mm precipitation resulted in runoff from the burned plot as compared to 23% of the same precipitation amount from the unburned plots. Soil loss from post-burned was 25 times greater. This study formulated a correlation of r=0.84 between sediment yield and percentage of bare soil. A similar finding is reported by O'Dea and Guertin (2003) in an area of southern Arizona grassland. The authors also recorded increased bulk density, runoff, and soil erosion on the one hand and lower plant cover, aggregate stability, and water intake on the other after burning grassland plots. A comparable conclusion of higher erosion and sediment yield was obtained by Wilson *et al* (2001) in New Mexico who compared pre-fire with post-fire scenarios by employing a GIS aided profile-based model called hillslope erosion model (HEM-GIS). However, prescribed and controlled burning well before the rainy season might have a positive effect on sustaining or improving the vegetative cover of an area and thereby reducing sediment production as observed in the montane grasslands of Natal (Everson *et al*, 1989).

2.4.4 Construction and roads

Construction sites pose very high risk of soil erosion (Hudson, 1995). Goldman *et al* (1986) reported that construction results in the most concentrated soil erosion that could reach from 2 to 40 000 times the undisturbed rate. Li *et al* (2001) examined the effect of wheels on infiltration under simulated rainfall. Their results concluded that traffic wheels reduced infiltration rate and time to ponding. The decrease in the rate of infiltration rate was found to be 4-5 times as compared to non-wheeled surface. Another study undertaken in Sam Mun, Thailand by Ziegler and Giambelluca (1997) to assess the impact of rural roads on runoff generation also recognized that the infiltration rate and hydraulic conductivity of unpaved roads were low leading to overland flow.

Nyssen *et al* (2002) recorded the effect of roads on gully formations in north Ethiopia. On a road segment of 6.5 km, 9 new immediate segments were created with additional seven further downslope. The researchers explained the reasons for the gullying as the road introduced concentrated runoff, increased catchment size, and diverted concentrated runoff to other catchments.

2.5 EFFECTS OF SOIL EROSION

Soil loss from an area not only results in reduction in depth of soil and nutrient pool but also has adverse effect on destination areas. In either case the costs of minimizing the problems are considerable. The impacts of soil degradation that eventually threaten human well-being are manifested as the loss of agricultural productivity and environmental quality (Lal, 1998b). Various approaches to evaluate the effects of soil erosion are followed depending largely on the purpose of the studies that are driven by the problems. In this study they are divided as on- and off-site effects that, in turn, summarize specific effects.

2.5.1 On-site effects

The on-site effects of soil erosion start by the loss of soil from an area as a consequence of which plant growth is hampered. The most important on-site effects are:

2.5.1.1 Loss of soil

As the term indicates, the first effect of soil erosion is loss of soil from source area (Troeh *et al*, 1980). So, in a local area consideration there is net soil loss due to soil erosion that can be manifested in a decrease in soil depth (Frye *et al*, 1985). Based on this, it is estimated that South Africa has lost 25% of its topsoil in the twentieth century alone (Laker, 2000). Adler (1985) puts South Africa's soil loss figure to about 10 ton per person per year. Specifically, Maud (1978) cited King's (1972) estimation of annual soil loss over the following ten years, based on the then prevailing conditions, to be 20 million tons. Miller and Gardiner (1998) reported that about 40% of topsoil has been lost in the last 100 years from the hills of Palouse of eastern Washington and adjacent areas in Idaho and Oregon. Likewise, De la Rosa *et al* (2000) estimated an average of 50 Mg ha^{-1} $year^{-1}$ of soil loss in Mediterranean experimental areas.

Lowery *et al* (1995) stated that soil losses have an impact on the soil physical properties and from 15 experimental sites in North Central USA, reported increases in bulk density and clay content and decreases in available water content and saturated hydraulic conductivity of the upper horizon.

2.5.1.2 Nutrient loss

Laker (2000) reports that South Africa loses nitrogen, phosphorus, and potassium due to soil erosion. An experiment in West Africa (Wild, 1993) showed 55 ton ha^{-1} of soil loss in 5 years carried away 1200 kg organic matter, 75 kg nitrogen, 16 kg phosphorus, and 34 kg exchangeable cations. In the USA 20 million metric tons of nitrogen, phosphorus, and potassium are lost with 13 billion metric tons of soil each year (Troeh *et al*, 1980).

2.5.1.3 Decline in productivity

As one of the effects of soil erosion, it is the decline in agricultural productivity that draws more attention (Pierce and Lal, 1994). Because soil erosion reduces organic matter, clay content, water retention capacity of soil, and plant rooting depth, it leads to poor plant growth affecting the overall biomass yield (Schertz *et al*, 1985). McCulloch and Stranack (1995) reduced soil depth progressively to examine the effect of rooting depth and found corresponding losses of sugar cane productivity on farms of the North Coast of KwaZulu-Natal, South Africa. In Alberta, Canada, Larney *et al* (2000) reported that a 20 cm topsoil removal resulted in 53% decline in grain productivity. Florchinger *et al* (2000) in Colombia, Pradhan *et al* (1997) in Canada, and Littleboy *et al* (1996) in Australia conducted studies and reported similar results. Such losses in productivity can be translated into cost. In the USA an Erosion Productivity Impact Calculator (EPIC) model was used and it was estimated that the annual soil loss amounts to $252 million (Crosson, 1998). According to the United Nations Environmental Program, UNEP (1991), about 20 million ha of land lose productivity to zero or are uneconomic annually because of soil erosion (Lal, 1994a). The same author has put that so far 430 million ha of productive land is lost, which is attributed to settled agriculture.

Continued reduction in productivity leads to cost of fertility sustenance (Morgan, 1995). In order to keep productivity at a required level, fertilizer has to replenish the nutrient shortage to sustain the fertility of the soil (Schertz *et al*, 1985). This cost increases with any increase in soil erosion. Magrath and Arens (1989, cited in Crosson (1998)) reported yield losses to be 4 to 7% per year, which is approximated to $323 million in a high erosion area of Java. In Mahaweli watershed, Sri Lanka, Gunatilake and Vieth (2000) used replacement cost method to study the on-site cost of soil erosion. The total cost of replacement was reported to be US $11.64 million per year out of which 76% was spent on replacing the nutrients, whereas the rest was used up to apply the fertilizer and repair the degraded fields.

2.5.2 Off-site effects

Awareness of soil erosion problem with respect to its effects off-site is drawing a growing attention. The damages of eroded soil out of the source area include flooding, sedimentation, and pollution as discussed herein.

2.5.2.1 Sedimentation

Soil transport continues until the steepness of the land decreases to the extent that renders the velocity of the water unable to entrain the soil particles (Troeh *et al*, 1980). This will be the point where deposition occurs which is undesirable in many cases. Clogging of irrigation canals and filling up of reservoirs and dams are results of sedimentation (Lee *et el*, 1998; Laker, 2000). Earlier, Perkins (1986) stated that 5.64 million m^3 of silt was trapped at the Driel Barrage on the Upper Tugela river in 10 years time. This amount was about 40% of the original storage capacity of the barrage. This river is estimated to carry seven million tons of soil annually owing to lack of conservation over large overgrazed areas (Anon, 1972, cited in Camp (1999)). The Wedbelt Dam in the Caledon River, Bloemfontein had a storage capacity of 114 million m^3 when it was completed in 1973 and after twenty years its storage capacity was reduced to 17 million m^3 due to sedimentation (Laker, 2000). Goudie (1981) reported that life expectancy of dams especially in the semi-arid areas of the USA was reduced to 30 years or less due to accelerated erosion. Because sediments reduce the storage capacity of dams, additional cost is required to clear the sediment after every depositional event. For example, removal of sediment from Cull Canyon Reservoir, California, 10 years after its construction cost about $1 million (Goldman *et al*, 1986).

2.5.2.2 Flooding

Soil displaced from its place by the action of running water inundates areas on its route. Flood, because of its soil content, has more devastating power than a runoff with less soil load. It is a usual experience to witness overland flow in extensively flat and steep areas. These actions can overflow and kill crops easily (Troeh *et al*, 1980). Laker (2000) reported that because of soil erosion even small rain events create flash floods in South Africa.

2.5.2.3 Pollution

Soils may enter dams or add up to soils of farmlands. If these soils contain chemicals and fertilizers, the deposition area will be polluted (Troeh *et al*, 1980). As quoted by Lal (1994a), Judson (1981) showed that sediments carried 10 billion Mg that drastically increased to 25 to 50 billion Mg after the introduction of intensive agriculture, grazing, and other activities. Damages by pollution should be overcome or minimized in the down slope area, hence adding extra cost (Troeh *et al*, 1980). In Elbe-River catchment area of Germany, a study showed that 11% of nitrogen input per year and 58% of phosphorus input per year flow into the surrounding water bodies (Frielinghaus and Schmidt, 1993).

2.6 THE NEED FOR MODELLING

Soil erosion by water cannot be avoided totally; rather it can be minimized (Wild, 1993; Hudson, 1995). Attaining the soil loss threshold is the objective of most conservation programs and this soil loss threshold is agreed to be in equilibrium with soil formation rate (Scotney and McPhee, 1992; Wild, 1993; Hudson, 1995; Miller and Gardiner, 1998). Soil loss tolerance is the maximum rate of soil erosion that still permits sustainable crop productivity (Wischmeier and Smith, 1978; Renard *et al*, 1997).

The first step in any conservation program is assessing the extent of soil erosion (Lal, 1994a), which could be quantitative in the case of available technology to process the existing data, or qualitative that depends mainly on visual observation. Quantitative expressions of soil erosion assessment that use empirical models are discussed here. The importance of models is immense for various reasons. In general models simulate the real phenomena by using a given small data and provide knowledge about the larger phenomena for the purpose of prediction (Mutchler *et al*, 1994; Miller and Gardiner, 1998). According to Nearing *et al* (1994, p.127), "a) erosion models can be used as predictive tools for assessing soil loss for conservation planning, project planning, soil erosion inventories, and for regulation; b) physically-based mathematical models can predict where and when erosion is occurring, thus helping the conservation planner target efforts to reduce erosion; c) models can be used as tools for understanding erosion processes and their interactions and for setting research priorities."

Covering the whole area to study soil loss is impossible; therefore it becomes necessary that a good representative site be taken to extrapolate its results (Young, 1994). The other advantage of models is related to its recommendation. The ultimate goal of soil erosion assessment is to conserve soil according to the nature and extent of the problem (Lal, 1994a; Young, 1994; Hudson, 1995). As empirical erosion models require data inputs, which are factors that affect soil erosion, as separate quantitative figures (Lal, 1994a), each input is treated for its share that indicates how important it is towards contributing to the overall soil loss. Consequently, a soil conservation program gives more consideration to factors that have greater share.

2.7 TYPES OF EROSION MODELS

Two main erosion model approaches are known: empirical and physically based or process-based model (Lal, 1994a; Hudson, 1995). An empirical model is based on observation/experiment, so it reflects observed facts and helps predict what will happen in the future. This model uses a database of given conditions making them applicable to those conditions. However, since there is no universally true model with any given conditions, it is important that each region has a model that matches its own condition and database. Examples of regional specific models are the Soil Loss Estimation Model for Southern Africa (SLEMSA) (Elwell, 1978) and the European Soil Erosion Model (EUROSEM) (Morgan *et al*, 1998).

A process-based model looks at each of the separate physical processes and then combines their effects. Because of the multiplicity of the variables involved in this category, it requires the use of computers (Hudson, 1995).

2.7.1 Process based models

The basic processes that initiated physically based models are detachment by raindrop splash, transportation by raindrop splash, detachment by surface runoff, and transportation by surface runoff. Later development of these models introduced the separation of rill from inter-rill erosion. The well known process-based models, namely EUROSEM, CREAMS, ANSWERS, WEPP are discussed here briefly. Also, a productivity model is summarized in this sub-section.

2.7.1.1 European Soil Erosion Model (EUROSEM)

This model is a product of the cooperation of European countries to develop European based soil erosion model to assess and predict soil erosion rates for designing conservation designs (Morgan *et al*, 1998). Realizing that annual soil erosion in many European countries usually results from few storms, this model was designed in a way that it assesses within storm events and sediment discharges. This model is similar to GUEST because of its involvement in single events and single slope segments and is also similar to the WEPP model for its evaluation of sediment concentration using experimental relationships (Rose, 1998). To satisfy the spatial variability of catchments by using dynamic distributed soil erosion modeling approach, EUROSEM is linked to KINEROS model, which has an advantage of water and sediment routing, and also to MIKE SHE model that is a continuous simulation model (Morgan *et al*, 1998).

EUROSEM gives specific attention to the effect of plant canopy on soil erosion through throughfall and leaf drips (Hudson, 1995). The components of this model can be defined as follows:

- Runoff generation has two components: surface depression and surface flow. Runoff is simulated as Hortonian, which represents surface flow as a result of infiltration-excess, and saturation flow expressed as depth.
- Soil detachment by raindrop is computed from the kinetic energy of the throughfall and the energy of leaf drip.
- Soil detachment by runoff is determined by comparing the estimated velocity of runoff with the critical velocity to detach, which is modeled as a function of grain shear velocity.
- Transport capacity is modeled as stream flow, which is computed as a product of flow velocity and slope.

Finally, erosion is compared between the amount of sediment available in the flow and the capacity of the flow (Hudson, 1995).

2.7.1.2 Chemicals, Runoff, and Erosion from Agricultural Management Systems (CREAMS)

CREAMS is mainly used to estimate the loss of pollutants from agricultural fields (Hudson, 1995). This model has three components: hydrological, chemical, and erosion (Hudson, 1995;

Morgan, 1995). The hydrology part estimates runoff volume and peak runoff rate, infiltration, evapotranspiration, soil water content, and percolation on a daily basis. The chemistry part estimates the nutrients and pesticides that are adsorbed and dissolved in the runoff, sediment, and percolated water. The erosion component includes empirical and process-based parts and estimates erosion and sediment yield in three steps.

- Overland flow that compares detachment by rill and inter-rill flow with transport capacity to estimate erosion and deposition rate. In this case erosivity and erodibility parameters are taken from the USLE.
- Channel flow that is used to estimate ephemeral gully erosion.
- Deposition that occurs as flow velocity decreases or due to ponding, which results from physical structures such as channel terrace.

2.7.1.3 Areal Non-point Source Watershed Environment Response Simulation (ANSWERS)

ANSWERS estimates sediment yield from the whole watershed (a non-point source) rather than from field-size (a point source) (Hudson, 1995). This model requires computers due to its large database. The model divides a watershed into elements of approximately 4 ha each having uniform input parameters. Then each element is assessed using four sub-models: hydrologic, sediment detachment, transport, and routing components such as overland, subsurface, and channel flow phases.

The erosion sub-model is partly based on an empirical model that uses some of factor relationships from the USLE. Basically, this model compares detachment capacity and transport capacity (Hudson, 1995).

2.7.1.4 Water Erosion Prediction Project (WEPP)

The processes this model considers are detachment, transport, and deposition (Morgan, 1995). This model is a new generation that is suitable for a wide range of personal computers. It contains three versions: a profile version, a watershed version, and a grid version (Morgan, 1995; Renard *et al*, 1996). The profile version computes the surface, soil, and crop properties related to the erosion processes. The watershed version takes the calculated sediment in the profile version and takes it through the channel system to the exit from the watershed. The grid version calculates the sediment delivery from each of the grids within an area. The limitation of WEPP is its difficulty to apply outside USA because of its requirement for a large data input (Rose, 1998).

2.7.1.5 Productivity models

A new approach is being introduced to estimate soil loss indirectly from change in productivity (Meyer *et al*, 1985). Erosion Productivity Impact Calculator (EPIC) is one example of models with such approach that combines empirical and physically based components. This model can estimate changes in productivity over long periods. Bernardos *et al* (2001) used this model to study agro-ecological change over 93 years period in Argentina. Assessing the effects of soil erosion on crop productivity is a difficult task since the relation between crops and soil is complex (Pierce and Lal, 1994). Moreover, productivity models, in general, require a large database and hence their applicability in developing countries needs to be further simplified (Lal, 1998a).

2.7.2 Empirical models

2.7.2.1 The Universal Soil Loss Equation (USLE)

The earliest method to assess soil loss used slope length and slope steepness, which was later, upgraded by including crop factor and mechanical protection for the first time (Morgan 1995; Renard *et al*, 1996). Browning *et al* (1947, cited in Wischmeier and Smith (1978)) studied the erodibility of soils and the effect of rotations and crop management on soil erosion. During the same time a team integrated previous studies with rainfall erosivity and soil erodibility (Morgan, 1995; Renard *et al*, 1996). The resulting equation developed by this group was called Musgrave Equation or Slope-Practice Equation due to the more attention given to slope and farming practice variables. This equation was given as Equation 2.1 (Hudson, 1995).

$$E = T \times S \times L \times P \times M \times R \tag{2.1}$$

where E is erosion; T is soil type; S is slope; L is length; P is agronomic practice; M is mechanical protection; and R is rainfall.

This equation was replaced by the Universal Soil Loss Equation (USLE) in the late fifties, which was a result of a database of more than 10 000 plot-years (Wischmeier and Smith, 1978; Hudson, 1995; Renard *et al*, 1996; Miller and Gardiner, 1998). The advantages of the USLE include:

- Consideration of variation in rainfall among storms or seasons
- Identification of localized climate and related crop management techniques
- Extension of the range of erodibility values due to results of experiments made on different soils
- Consideration of inter-relationships within crop management such as level of productivity, crop sequence in the rotation and the management of residues.

The USLE serves specific purpose in that by estimating the long-term average annual soil loss, it allows farmers and conservationists to select the best conservation method that keeps soil loss at tolerable limit (Wischmeier and Smith, 1978). Although this equation was originally meant for use on cropland it has been tested and found useful in other areas, too (Troeh *et al*, 1980). However, USLE has limitations some of which are that the model:

- doesn't predict sediment yield
- doesn't predict soil loss from a single storm since the factors are long term averages
- doesn't predict soil loss outside its range (example, slope is limited up to 16%)
- doesn't predict soil loss from large rills and gullies

The USLE is designed to include factors that affect soil erosion, and hence the following equation (Wischmeier and Smith, 1978):

$$A = R \times K \times LS \times C \times P \qquad (2.2)$$

where A is the average annual soil loss (ton ha^{-1} $year^{-1}$); R is a measure of the erosive forces of rainfall and runoff (MJ mm ha^{-1} hr^{-1} $year^{-1}$); K is the soil erodibility factor (ton ha hr ha^{-1} MJ^{-1} mm^{-1}); L is the slope length factor; S is the slope steepness factor; C is the crop management factor; and P is the conservation practice factor.

In the above equation L, S, C, and P are dimensionless ratios that compare the site under observation with standard conditions. Whereas, R is the sum of each erosivity energy expressed in MJ ha^{-1} multiplied by the maximum rainfall amount measured for 30 minutes in mm hr^{-1}.

K is the average soil loss in tons per hectare. K values are given for most soils in tabular form, and hence they can readily be read. However, this nomograph cannot represent every soil type since it is developed for the soils of the USA; therefore its transferability to other countries is limited (Hudson, 1995).

2.7.2.2 Soil Loss Estimation Model for Southern Africa (SLEMSA)

Since the USLE is developed in the USA, it needs to be modified elsewhere to include different situations such as steep land, tropical crops or soil properties like organic matter. Also, the USA has rich database to feed the USLE requirements fully, which might be inapplicable in other data-poor countries. Therefore, models like the SLEMSA that require small data inputs are needed (Elwell, 1978).

Elwell (1978) considers five parameters in the SLEMSA model: seasonal rainfall energy (E), the amount of rainfall energy intercepted by the crop (i), soil erodibility (F), slope length (L), and slope % (S). These factors are rearranged to give three principal sub models known as, a sub model to estimate soil loss from bare soil, a sub model to account for cropping practices, and a sub model to account for topographic differences. The equation of this model is:

$$Z = K \times C \times X \qquad (2.3)$$

where Z is the predicted mean annual soil loss (ton ha^{-1} $year^{-1}$); K is the mean annual soil loss (ton ha^{-1} $year^{-1}$) from a standard conventionally-tilled field plot of 30 by 30 m at a 4.5% slope, for a soil of known erodibility, F, under a weed-free bare fallow; X is the ratio of soil loss from a field slope of length L (m) and slope %, to that lost from a standard plot; and C is the ratio of soil loss from a cropped plot to that from bare fallow. The SLEMSA model, apart from its use with limited data, allows for improvements as more data are obtained.

This model is closely related to the USLE in that both estimate long-term average annual soil loss and both compute cover and rainfall energy for a 15-day period (Hudson, 1995). For example, this model was applied in the Drakensberg area to assess annual soil erosion. The result showed that that soil loss range is 1 to 20 ton ha^{-1} $year^{-1}$. Moreover, the results obtained from 266 grid points were extrapolated to prepare an erosion map for the large area (Scotney, 1978).

Hudson (1995) also differentiates SLEMSA form USLE in that:

- SLEMSA doesn't use P factor of the USLE because it can be included in L or S factors, or within the erodibility F factor.
- SLEMSA uses E (instead of R in USLE) and measures the total annual kinetic energy of the rainfall, which is easier to find from rainfall records than the EI index.
- C in SLEMSA is determined from the density of crop cover taken in the field at 10-day intervals over 180-day growing season. The overall C is then the ratio of soil loss from cropped plot to that of a bare fallow. This model is also used on rangeland with slightly different sub-model to relate C to cover density.
- SLEMSA instead of K uses F (ton ha^{-1} $year^{-1}$) and is based on soil type.
- X is used in SLEMSA as topographic factor with very slightly different equation of LS factor of the USLE.

2.7.2.3 SOILOSS

This model uses the same factors as the USLE, but developed to suit the local experience of Australia by modifying the factors R, K, and C (Hudson, 1995). In this model the USLE EI_{30} erosivity index is replaced by a more process based QE_A, where Q is the runoff rate and E_A defines the rate of kinetic energy. The USLE nomograph uses particle size distribution derived from laboratory experiment by shaking a soil sample using a chemical dispersant, which was believed to have a more exaggerated result than the reality. In contrast, shaking a sample using only water gave a result of stronger correlation with field measurements.

Crop management factor C in SOILOSS model has the principle of C similar to the one in the RUSLE but was determined in Australia using simple plant growth, tillage, and residue decay models.

This model in general uses a computer program as the RUSLE. Further, it is applicable to urban land, pasture, and rangeland and has room to accommodate new introductions in RUSLE.

2.7.2.4 The Revised Universal Soil Loss Equation (RUSLE)

The Revised Universal Soil Loss Equation (RUSLE) released in the early 1990's is meant to overcome the limitations of the USLE, but still inherits its basic properties from the USLE (Renard *et al*, 1994; Renard *et al*, 1996; Renard *et al*, 1997). One of the most important advantages of the RUSLE is its use of computer program (Renard *et al*, 1994; Renard *et al*, 1996; Renard *et al*, 1997; Miller and Gardiner, 1998). Developments that USLE could not accommodate and hence prompted its revision include the following (Hudson, 1995):

- introduction of new farming practices such as ridge-tillage and strip-cropping,
- inability to account runoff reduction practices like channel terraces,
- inability to estimate erosion of ephemeral gullies, and
- inability to estimate erosion from rangeland

As stated above in its retaining of the basic properties of the USLE, the RUSLE uses the same skeleton of equation (Equation 2.2) as the USLE. The modifications of each factor are summarized below.

2.7.2.4.1 Rainfall erosivity (R)

One of the changes in this factor is to reduce its value in the case of flat areas, and long and intense rainstorms (Renard *et al*, 1994; Hudson, 1995; Renard *et al*, 1996; Miller and Gardiner, 1998). The reason for this is water ponds reduce rain erosivity because of their shield effect of the soil against raindrops.

2.7.2.4.2 Erodibility factor (K)

The nomograph used in the USLE is retained, but erodibility data from the world have been reviewed to develop a new equation that estimates K as a function of diameter of soil particles (Renard *et al*, 1994; Hudson, 1995; Renard *et al*, 1996). Renard *et al* (1994) have reported the advantage of the K factor in the RUSLE to vary seasonally. In addition, the RUSLE has incorporated the temporal variability of soil erodibility that is related to soil moisture and soil temperature changes (Miller and Gardiner, 1998). These authors put the difference of K values for the USLE and the RUSLE to be more than 20%.

2.7.2.4.3 Topographic factors (LS)

The revision of these factors resulted in the reduction of soil loss (Hudson, 1995). The RUSLE LS factors also consider differences in the ratio of rill to inter-rill erosion (Hudson, 1995; Renard *et al*, 1997). Of tremendous improvement, LS in the RUSLE considers segments on the slope to accommodate the non-uniformity of slopes as compared to the USLE that doesn't consider local variations of a slope (Renard and Ferreira, 1993; Renard *et al*, 1997). Miller and Gardiner (1998) estimated that the difference of the LS factor between the USLE and the RUSLE would go as high as 57%. In the same way, Renard and Ferreira (1993) reported that RUSLE-computed soil loss on steep lands is half that computed by USLE.

2.7.2.4.4 Crop cover factor (C)

Unlike the USLE that takes C values from a table of most frequent crop culture, the RUSLE calculates the C values from five sub-factors (Renard *et al*, 1997) listed below. The C factors are similar for both equations except in the case of conservation and no-till conditions where C factor is lower for the RUSLE. This overestimation of C factor in the USLE for no-till may reach 300%.

- PLU, the prior land use factor
- CC, the canopy sub-factor, which reflects the effectiveness of vegetation cover in reducing rainfall energy
- SC, the surface cover factor that looks at the effect of ground cover
- SR, the surface roughness factor
- SM, the soil moisture subfactor

Crop cover factor (C) is then calculated over a 15-day period and combined with 15-day values of R and K. Moreover, the RUSLE can be applied to crops for which there are no experimental data.

2.7.2.4.5 Support practice factor (P)

This factor is the most uncertain in both the RUSLE and the USLE, because different studies show variations in their results (Renard *et al*, 1994; Hudson, 1995; Renard *et al*, 1996). However, the P factor in the RUSLE considers more variables than it does in the USLE (Renard *et al*, 1994; Renard *et al*, 1996; Miller and Gardiner, 1998). After evaluating extensive data to assess the effect of contouring on soil erosion, factor values such as slope, ridge height, row grade, and climatic erosivity are identified as decisive parameters to consider. Strip cropping consideration in the RUSLE includes sediment movement both to the strips and through the strips (Miller and Gardiner, 1998).

2.8 REMOTE SENSING, GIS, AND EROSION MODELS

2.8.1 Application of remote sensing

Remote sensing refers to the acquisition of data about an object by a sensor without direct physical contact with it (De Jong, 1994a; Lillesand and Kiefer, 1994). Application of remote sensing to natural resources inventory has become fundamental for management that is friendly to the environment (Lillesand and Kiefer, 1994). It gives information about the physical characteristics and variation of features on the ground that can be converted into image for interpretation and analysis. This variation of land features, which reflects the inherent differences of their build-ups, is sensed and recorded by the remote sensor that receives the electromagnetic response of the features (Lillesand and Kiefer, 1994; Nizeyimana and Petersen, 1998). Such capability to differentiate features is being well applied in soil survey. Remote sensing classifies soils according to their chemical and physical differences that are readily sensed by remote sensors (Nizeyimana and Petersen, 1998; Ritchie, 2000). Such variations are most likely due to soil moisture, organic matter, mineral content, and others (Lillesand and Kiefer, 1994).

Weismiller *et al* (1985) summarized studies that showed the relationship of reflectance with soil erosion. Most of these studies have indicated that organic matter and iron oxide have strong effect on reflectance characteristics. Collins *et al* (1998) suggested that indirect relationship could be made between soil reflectance properties and soil structure. This can be explained by the fact that the structure of clay soils improves the moisture holding capacity that has low reflectance. In contrast, sandy soils have low water holding capacity, which has high reflectance.

Remote sensing can be applied directly or indirectly to assess soil erosion (Pelletier, 1985; Ritchie, 2000). Photo interpretation and photogrammetry are used to directly identify eroded areas from images or images can be used as inputs in GIS. For example, crop cover or rainfall interception by vegetation can be assessed from remotely sensed images (De Jong *et al*, 1999). Garland and Broderick (1992) used aerial photos of 1944/5 and from 1976-81 to assess soil erosion change in the Tugela catchment of KwaZulu-Natal in which they found a decline in the problem. Randall (1993) delineated gully and sheet erosion areas using Landsat TM images in Olifants River catchment, South Africa to find that gullies could be mapped more accurately. Similarly, Liggit (1988) used remotely sensed data to assess soil erosion in Mfolozi. Liggit (1988) interpreted orthophotos and aerial photographs taken at different times and scales to estimate the spatial extent of gullies and sheet erosion.

Delineation of gullies using aerial photos was also undertaken in Spain by Garg and Harrison (1992) by using their differences in tone and texture from the surrounding land surfaces. Hochschild *et al* (2003) used aerial photographs of 1990 at 1:30 000 scale and Landsat 5 TM image of 1996 in their assessment of soil erosion in Mbuluzi river catchment (Kingdom of Swaziland). The study mapped existing erosion types and extents by creating negative correlation between the erosion levels and the percentage of vegetation cover. The satellite image was used to derive the vegetation cover density and the R(USLE) C-factor using the vegetation indices and leaf area index. A similar technique was applied in the upper Mkomazi river catchment of KwaZulu-Natal, South Africa by Flugel *et al* (2003). In Nsikazi, Mpumalanga Province of South Africa, Wentzel (2002) used satellite imagery to derive bare soil index for likely soil erosion mapping.

Daba *et al* (2003) employed remote sensing technique to directly assess gully erosion by measuring gully erosion change in eastern Ethiopia using aerial photographs taken in 1966 and 1996, both at 1:45 000 scales. The technique used the break lines and constructed straight lines to create closed polygons from which volumes were calculated using cut and fill system. The result showed soil loss of 1.7 ton m^{-2} over the 30 years time period. The break line measurement was found to be in agreement with the field measurements of the gully depth and width. Similarly, Khan and Islam (2003) identified a great deal of riverside retreat by using time series analysis of satellite images in Brahmaputra-Jamuna River, Bangladesh.

Researchers such as Sayago (1986) applied remote sensing in conjunction with field measurement of erosion using the USLE to map erosion hazard in the northwest of Argentina, where multi-temporal Landsat images and conventional aerial photos were used to delineate mapping units. This study, which, in general, produced a map of high erosion risk in the area, has concluded that remote sensing is efficient in detecting erosion hazard areas economically and within a short period of time. Earlier, Morgan *et al* (1978) examined the advantage of remotely sensed data in soil erosion assessment in Wisconsin, USA. They estimated soil loss using the USLE by entering data obtained from aerial photographs. The predicted soil loss figures were found closely comparable with the results derived from field-gathered data.

The advantage of remote sensing in extracting information from a larger area and much quicker than the conventional survey systems is acknowledged by several researchers including Morgan *et al* (1978) and Engman (1996). Time scale is an important consideration that helps gather repetitive data from the same space. In addition to its wide area coverage, remote sensing also provides areal data as opposed to point data made by sampling in conventional surveys (Baumgartner and Apel, 1996; Engman, 1996).

2.8.2 Application of Geographic Information Systems (GIS)

A GIS is a system that deals with georeferenced spatial data to perform such processes as capturing, storing, retrieving, manipulating/analyzing, and displaying the output of such spatial data (Burrough, 1986). A widely accepted definition of GIS is that it is a computerized system to process spatially related information (Collet *et al*, 1996). This description includes three portions, namely, the computer hardware integrated with the software component, processing of information, and the spatial nature of the information.

Recent developments of Digital Elevation Models (DEMs), soil databases, remotely sensed data, and the related computers performance to process large data sets have increased the role of GISs in natural resource assessment (Petersen *et al*, 1998). According to Zhu *et al* (2001) mapping of soil landscape showing detailed spatial and attribute anomalies is possible with more regular updates and less cost compared to the conventional soil survey technique.

De Roo (1996) pointed out two advantages of using GIS in soil erosion models. First, it resolves spatially variable runoff and soil erosion process through generation of cells to account for the variations. Second, GIS is capable of inputting large data automatically, which is difficult to enter manually. Such quality of GIS has also helped develop more detailed distributed erosion and deposition models (Lorentz and Howe, 1995; Jetten *et al*, 2003). Mongkolsawat *et al* (2002) based on a study in Thailand reported the advantage of GIS in analyzing multi-layer data spatially and quantitatively. GIS is also credited to its capability of assessing soil erosion at large scale (Wijesekera and Samarakoon, 2001; Lufafa *et al*, 2003). It also has become possible to readily estimate soil loss from existing digital data sets of soil map, terrain data, and remotely sensed data (Petersen *et al*, 1998). More important, since the conventional surveys require extensive manpower, time, and cost, GIS has become an excellent tool to perform environmental studies (Marble, 1987).

GIS is also being used with soil erosion assessment models. Brazier *et al* (2001), for their study in Cambridge, UK, used GIS with MIRSED (minimum requirement version of WEPP) to assess hillslope scale soil erosion rates. Similarly, De Jong *et al* (1999) applied the Soil Erosion Model for Mediterranean areas (SEMMED) that integrated Landsat TM images, a digital terrain model, a digital soil map, and soil physical field data to assess soil erosion at regional scale in southern France and in Sicily.

Linkage between GIS and a soil erosion model depends on both the GIS and the model itself. GIS and grid cell-based models can easily be combined because of the capacity of GIS to analyze raster data (Petersen *et al*, 1998). It is easy to interface empirical models such as the RUSLE with GIS because of the simplicity of the equation (Ferro *et al*, 2001), whereas process-based models require detailed data input to make multiple layers (Petersen *et al*, 1998).

2.9 CASE STUDIES

Application of remote sensing and GIS in soil erosion assessment is being adopted faster despite its recent advent. Repeatedly said, the integration of GIS and remote sensing techniques has an advantage to assess soil erosion at a large spatial and temporal scale. This section looks at only a number of selected researches on soil erosion assessment around the world that involve the use of GIS, remote sensing, and USLE /RUSLE/MUSLE.

The Upper Ewaso Ng'iro basin of Kenya is one example that used the USLE (older version of RUSLE) and GIS to assess erosion hazard (Mati *et al*, 2000). The area covered in the study is 15 251 km^2, which is too large an area to be covered with detailed experimental observation. Detailed information about this study revealed that data were obtained from weather stations, reconnaissance surveys, topographic maps, and plots. Each factor of erosion was changed into a layer of grid cells of 100 by 100 m using ARC/INFO GIS. The USLE was then applied to each cell of approximately 1 ha. In deriving each factor, rainfall from six stations was used to calculate storm erosivity indices for a number of years for each station.

To find soil erodibility, the soil map of the basin was digitized from the available Exploratory Soil Map of Kenya at a scale of 1:1 000 000. Basic data were obtained from laboratory analyses of 83 representative soil samples to determine % sand, silt, clay, and organic matter, which were used to estimate erodibility using the USLE soil erodibility nomograph. The digitizing resulted in 36 soil map units contained in the study area. Then the soil map was reclassified based on the mean K factor of each soil type.

To determine the topographic factors, 32 sheets of topographic maps at 1:50 000 scale were digitized and the DEM was built from the contours and the drainage system. Using ARC/INFO GRID the slope steepness and slope length was determined from the DEM.

The crop and management factor was determined from land cover map prepared using remote sensing techniques and field surveys to ground truth the map. The USLE guide tables for C factors were used to reclassify the land cover map to find the C-factor of the area. Conservation practice (P) was obtained using the maps of the area that were conserved. These areas were digitized and in combination with the annual reports were used to summarize the information on the types of conservation structures. Finally, the factors were multiplied using the USLE equation in ARC/INFO GRID. The result produced five classes of soil erosion. This study was useful because it made possible to quantify soil erosion that covered a very large area. In addition to this, the technique helped to interpolate observation from data rich to data poor areas. Moreover, the results obtained were comparable to soil loss estimation from plots. However, the study also had limitations one of which was the scale of the data that was small. This created problem in determining reliable values of topographic, crop, and conservation management practice factors. The second drawback was related to the P factor where appropriate value for local conservation practices could not be determined.

The case study at Zenzontla sub-catchment employed the RUSLE integrated with a GIS (Millward and Mersey, 1999). This research had an objective of modeling soil erosion potential for soil conservation planning. The materials used in this study were rainfall records from 30 stations around the sub-catchment, topographic map at 1:50 000 scale, soil map at 1:50 000 scale, and a full Landsat TM scene. In deriving the rainfall-runoff, R, factor, since all the weather stations were outside the study area, interpolation was made in IDRISI. Then to derive the RUSLE R factor EI_{30} index was used. The topographic factor, LS, was extracted from a DEM, which was generated by digitizing 20-m contour lines from the topographic map. The erodibility, K, factor was determined from the soil map which utilized the UNESCO-FAO classification system. After the soil map was digitized to extract the soil data only for the sub-catchment, values were assigned that correspond to the soil types. The digital land cover map was used to derive the cover management, C, factor. The scene was first rubber sheeted to match ground control locations and then the area defined by the sub-catchment was extracted that covered 112 km^2. The newly extracted image was then classified by supervised classification using maximum likelihood classifier. The bands used for the classification were 1, 3, 4, 5, and 7. The support practice, P, factor that refers to the effects of contouring and tillage practices is given the maximum value of 1 since there were no conservation practices. The resulting soil loss prediction was then given in 5 classes: minimum, low, moderate, high, and extreme with their numeric range values. The output was verified against field observation and comparison with the Global Assessment of Soil Degradation (GLASOD) maps to appreciate the model's predictive accuracy. The field verification proved that the RUSLE model's ability to predict soil erosion in the Zenzontla sub-catchment is very good. Comparing the model with the GLASOD map, it is found that the model results are equally characterized by the GLASOD classification.

In an area south east of Lake Tana, Ethiopia, image of Modular Optoelectronic Multispectral Stereo-Scanner (MOMS-02/D2) taken in 1993 at 307 km orbit altitude was used to quantify and map soil loss rates using USLE (Reusing *et al*, 2000). The image combined multispectral bands 3 and 4, and panchromatic stereo-bands 6 and 7. This image was used to delineate different land use classes following a supervised classification of the multispectral bands to assign respective C factor values, whereas the panchromatic bands were used to develop Digital Terrain Model (DTM) of the area from which the topographic factor (LS) was extracted. The composite color of this image was also used to map soil types where soil surface is exposed and extrapolation was used to map soil types in vegetated areas. These soils were then digitized and transformed in ARC/INFO grid format to be later mapped according to their K values. The final soil erosion map was calculated in ARC/INFO by combining the factor maps. This result showed that the majority of the 75 km^2 study area experienced soil loss rates of 256 ton ha^{-1} $year^{-1}$. The research concluded that high-resolution remote sensing data could be a suitable and affordable means for natural resource management. Moreover, use of one remotely sensed image data set to extract several variables had envisaged its importance in future studies.

Another case looks at the application of GIS and remote sensing in the Indian Bata River Basin to develop a soil erosion intensity map using MUSLE and to prioritize the watershed based on the severity of soil erosion (Mohamed Rinos, 2002). The area covers about 268.68 km^2. The study used remote sensing data, namely, IRS-1DLISSIII on different dates: 12 October 1998 and 01 March 1998 and IRS-1DPAN taken on 08 October 1999, the path and row for all cases being 96 and 50, respectively. The remote sensing data were used to identify land use/land cover of the area. Supervised classification using maximum likelihood classifier was used to classify the area into 8 classes. The classified land use/land cover map, which was also supported by ground truthing, was used to prepare the C factor. In addition, SOI toposheets taken in 1965 at a scale of 1:50 000 were used. From these sheets DEM was created by digitizing contour lines and spot heights. The DEM was used to derive the slope and aspect maps. These maps were used to extract the topographic, LS, factor. The other data used were pedological map and soil map to generate the erodibility, K, factor. The P factor values were chosen from previous research findings. The meteorological data obtained from 4 stations included records of 2 years to 20 years. The final erosion intensity map of this area, obtained by multiplying all the above factors as defined by the MUSLE, was classified into five. This research has concluded that MUSLE is an efficient model to assess soil erosion intensity for conservation management. It was also made possible to provide valuable data on C and P factors. Moreover, the use of GIS was helpful to compile and analyze data within a short period of time at very high resolution.

Mongkolsawat *et al* (2002) also tested USLE and GIS to map soil erosion in Thailand with specific objectives of establishing spatial information of soil erosion with the USLE and evaluating the utility of GIS in soil erosion mapping. The study area, Huai Sua Ten watersheds, covers about 41 080 ha. The inputs used for this study were rainfall data from 11 years of records, soil map of Thailand, topographic maps, and Landsat TM imagery. Rainfall-runoff erosivity factor was computed from the rainfall data. The soil map of Thailand was used to generate the erodibility, K, factor. Elevation and contour maps of topographic map at 1:50 000 scale were overlaid to produce a polygonal layer which was, in turn, used to derive the topographic, LS, factor. C factor was extracted from the Landsat TM imagery. The P factor was determined to be 0.027 for paddy field as compared to 1 for all other vegetative cover types, which had no erosion control. Following the determination of the USLE factors using the combination of remote sensing, GIS, and USLE equations, each factor was encoded in a GIS environment to produce five thematic layers. Spatial overlay of the USLE factors of the area gave a soil erosion map of eight classes. The study gave a general picture to show the causes of soil erosion spatially and quantitatively. In addition, the remote sensing and GIS technology produced data for decision support system. Comparing the results of this study with previous study, using USLE showed the presence of variation in the values that could apparently be due to the topographic and cover factor.

Erosion assessment at the Romero River Watershed using remote sensing and GIS is also another good example that showed the technology is an excellent tool for assessing land degradation, land use changes, and soil and water resource changes over time (Perlado, 2002). Its objectives were to generate spatial database using geographic information systems and remote sensing and to make erosion estimation in the area. The erodibility, K, factor was determined from the laboratory analysis of soil samples. The topographic, LS, factor was computed from 4 sheets of topographic maps at 1:50 000 scale. These maps were scanned and then imported into the GIS software where they were georeferenced and mosaicked. The vegetation cover, C, factor was estimated based on previous experiments done using ^{137}Cs in the watershed. Moreover, soil productivity study is being undertaken using the GIS technology. The result of this study helped show critical areas of concern. In general, the combination of remote sensing and GIS was found to be excellent for monitoring land degradation, land use changes, and soil and water resource changes over time.

A study in Kegalle district of Sri Lanka also stresses the significance of GIS in modeling soil erosion using USLE over large spatial extents. The study area covered 23 000 ha. This raster-based approach used a square cell of 25 by 25 m, whose accuracy is considered reasonable given the original data taken from 1:50 000 maps. Four data layers, with each having equal cell size, were overlaid to give the final annual soil loss at each cell. From the model, the average soil loss was found to be 29.7 ton ha^{-1} $year^{-1}$ as compared to the acceptable soil loss tolerance range of 2.5-12 ton ha^{-1} $year^{-1}$ as given in the literature (Wijesekera & Samarakoon, 2001).

Cox and Madramootoo (1998) applied the RUSLE model in a GIS environment with the objective of evaluating agricultural land management practices based on potential soil loss in St. Lucia, Eastern Caribbean with area coverage of 616 km^2. The study used DEM, a three-dimensional raster representation of the topography, soil type, and land use coverage. These input data were used to derive slope length and slope steepness factors from the DEM, soil erodibility factor from the soil type coverage, and crop management and conservation practice factors from the land use coverage. The rainfall erosivity factor was extracted from previously developed isoerodent map for the area. These factor layers were then multiplied according to the RUSLE model. Results were given in different scenarios/simulations to represent the effect of conservation practices under the current land use systems and predicted land uses. Generally, this study has given a basis for further study in the field and prioritizes areas that need remedial actions. It also revealed GIS's capability to help plan watershed management in the area. This study also recommended for modification of the topographic factors especially on the steep lands of St. Lucia that gave high soil loss figures. A similar approach to assess the worst case scenario of soil erosion was done in Langkawi Island, Malaysia using remote sensing, GIS and USLE (Baban and Wan Yusof, 2001).

Ogawa *et al* (2002) applied remote sensing, GIS, and USLE to make similar study in Pakistan with an area covering 27 500 by 23 000 m. Rainfall data were used from the records of test site location. The rainfall, R, factor was obtained by multiplying storm kinetic energy with the maximum 30-minute storm intensity. Soil erodibility, K, factor was taken from a table that was prepared to show correlation between soil characteristics and erodibility estimations based on previous experiments. One of the topographic factors, slope steepness, was derived from DTM for each pixel. The DTM, in turn, was prepared from a topographic map at 1:50 000 scale in ARC/INFO. The slope length was fixed to be 22.13 m because calculating from DTM could create very long length. The two sub factors were then combined to get the final topographic, LS, factor. The vegetation cover, C, factor was referenced from a table following a supervised classification of Landsat TM image. Similarly, land classification map and another reference table for erosion control practice factor were used to determine the conservation practice, P, factor. Finally 6 classes of soil loss level were shown. The method enabled to estimate soil erosion for a large area. In addition, the method was useful to detect land use and environmental change. The limitations of the study were reported to be the inability to get more accurate data and lack of detailed field survey.

In South Africa, the RUSLE model was used within the Agricultural Catchments Research Unit (ACRU) hydrological modelling system by Kienzle *et al* (1997). The study was done in Mgeni Catchment, KwaZulu-Natal with an area of 4078.5 km^2. Because of its large size, the catchment was divided into 250 by 250 m grid cells for each of which the RUSLE factors were computed. The rainfall erosivity factor was developed for the area that had correlation to the EI_{30}. The soil erodibility factor for each grid was generated by overlaying the terrain unit map, which itself was prepared by combining slope, surface curvature, and flow accumulation information, with the existing Land Type coverage that has soils information for each terrain unit. A DEM of 250 by 250 m was used to calculate the slope steepness whereas techniques that combined slopes, flow accumulation, and algorithms developed on GIS helped calculate slope length. The cover factor for each cover type identified in the area was calculated and incorporated into GIS coverage. The conservation practices factor was obtained by combining land cover with slope information and then rules were assigned for land use types in the study area. Out of the above factors, the erosivity and the cover factors were prepared on monthly bases. By overlaying the factor maps monthly soil loss potential maps and annual potential map were created. Among other advantages such as displaying of soil erosion spatially at different risk level, this work indicated that estimating factors and consequently soil loss on monthly bases first and adding to get the annual soil loss is preferable because of temporal variation of rainfall and vegetal cover.

2.10 SUMMARY

Based on the period of occurrence of the problem, soil erosion is divided as geologic, that is driven by natural processes, and accelerated, which is aggravated by human activities. The problem, which is manifested in different forms, is influenced mainly by climatic, soil condition, topography, and management activities. Correspondingly, several soil erosion assessment tools that have come as a result of the ever-increasing consequences of the problem, take these most important driving variables as their inputs. Though, recently, models that depict nearly all the processes involved in soil erosion from the detachment to the deposition are being developed drawing increasing interest of hydrological studies, empirical models are still enjoying wide applications. Of these, the RUSLE model is the most widely used and is well integrated with GIS and remote sensing technologies. Because of the ease in acquiring, storing, and analysis of large spatial data, many studies have made use of such integration with reported successes with regard to time and cost saving, and accuracy of results. Using similar method, this study aims at assessing in detail the soil erosion condition of WGR whose description is given in the next Chapter.

CHAPTER THREE: STUDY AREA

In literature, Weenen Game Reserve (WGR) is also known as Weenen Nature Reserve (WNR). In this thesis the name is taken from the official visitors guide and recent literature. This chapter briefly describes the study area. The chapter focuses mainly on the physical, climatic, and biotic nature of the area. However, the socio-economic activities are presented in limited details principally due to inadequate documentation. The lack of documentation might be due to the fact that settlement even in the past had not been permanent.

3.1 GEOGRAPHICAL SETTING AND SIZE

The geographical location of Weenen Game Reserve (WGR) is 28°49'-28°56'S and 29°57'-30°03'E (Figure 3.1). It is about 25 km northeast of Estcourt, and 8 km to the west of Weenen Town, in the midlands of KwaZulu-Natal (Camp, 2001). WGR is bounded by Weenen Town lands to the northeast, and grazing and some cultivation lands on all other sides (Bourquin and Mathias, 1995). The total area of the reserve is approximately 4891 ha and is physically divided into north and south by a steep valley through which the Bushman's River flows. The northern section covers the larger portion and has an area of about 3153 ha.

3.2 HISTORICAL BACKGROUND

In the past Weenen Game Reserve (WGR) consisted of two sections, namely, Onverwacht and Boesmans River Poort (Camp, 2001). The author also summarized the history of the reserve based on its ownership chronology as given in Table 3.1.

People who resided in the area were tenants who worked for free on farms elsewhere during farm season. During their off-farm season stay in the area, the tenants had unlimited access to cattle grazing leading to severe land degradation of the area. As a result of this the area was expropriated by the State to serve as research area run by the Department of Agriculture with specific objective of testing soil conservation techniques to fight soil erosion problem.

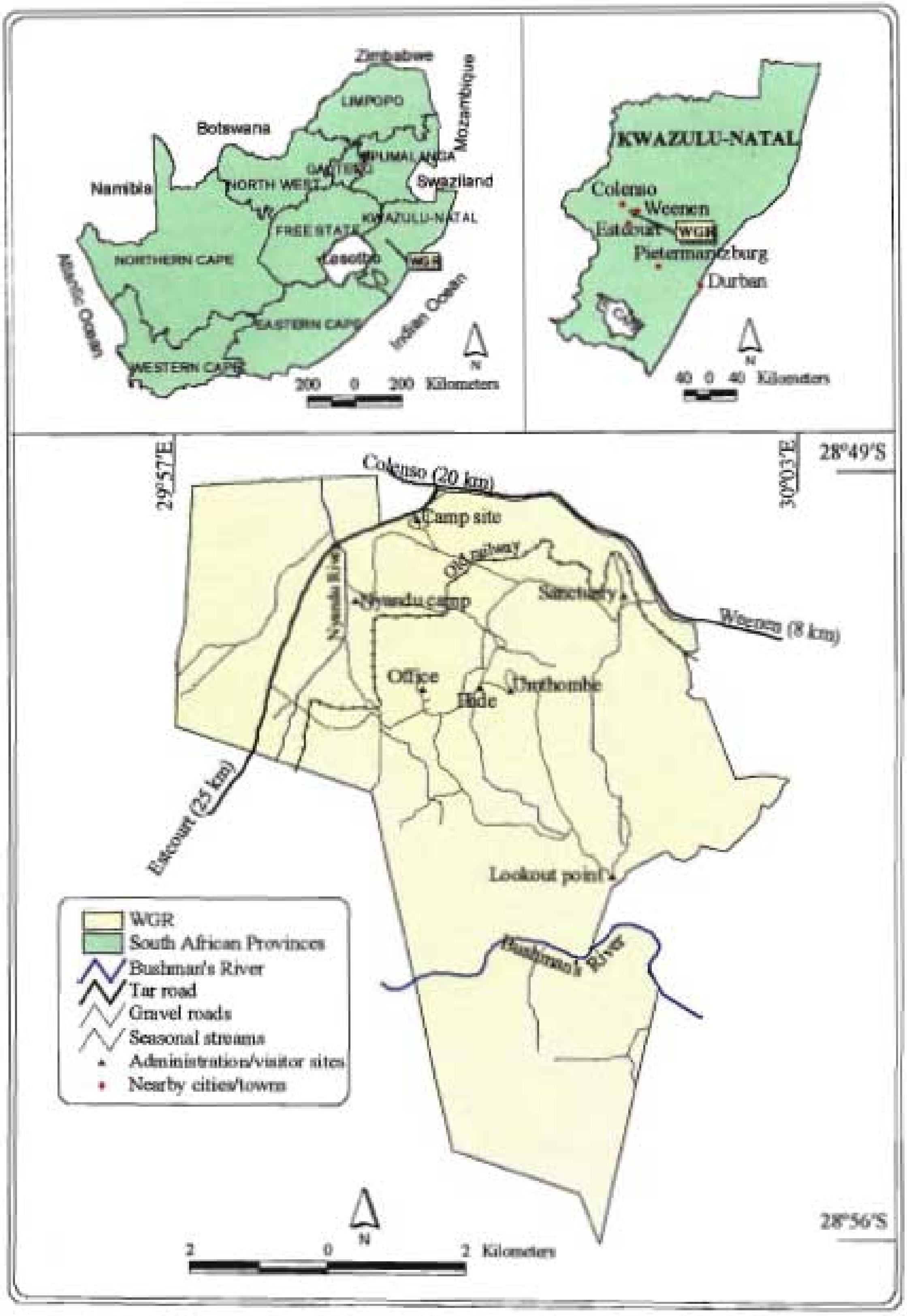

Figure 3.1 *WGR and its location map*

Table 3.1 *Historical ownership succession of WGR* (Camp, 2001)

Period	Ownership
1830 – 1862	Coetzee family
1862 – 1875	Andries Pretorius, the leader of Voortrekkers in the area
1875 – 1888	Mkungu
1888 – 1892	Theophilus Shepstone, diplomatic agent to the "native tribes" in Natal
1892– No date	J Geekie, the owner who initially used Onverwacht as a "labour farm"
1944 – 1948	The period of expropriation of the area

In 1975 the area was passed over to KwaZulu-Natal Nature Conservation Service (KZNNCS), a body that has been running the reserve ever since (Camp, 2001).

3.3 CLIMATE

WGR is well known for its experience of erratic rainfall. Edwards (1967) reported that the area is also characterized by high intensity rainfall. The annual precipitation of the reserve based on the records from 1981 to 2002, ranges from 489 to 1073 mm with an average of 724 mm and standard deviation of 152 mm. The area receives summer rains and the main rainy months of the reserve run from October to March.

Breebaart *et al* (2001) described the temperature of the area during summer as hot with daily maximum average temperature of 30°C, which could go as high as 37°C in January. On the other extreme the winter months, characterized by little rainfall, have a mean temperature of 10°C and a minimum record of less than 0°C.

3.4 GEOLOGY AND SOILS

WGR lies on shales, mudstones and sandstones of the Beaufort Series (Karoo System) (Hughes, 1989; Bourquin and Mathias, 1995). Dolerites plateaus are seen in the south and southeast of this system. The northern and western portions of the reserve, which are based on Beaufort Shales, are relatively flat (Hughes, 1989).

The prominent characteristics of the soils in the area are shallowness and erodibility (Hughes, 1989; Ghebremeskel, 2003), the main reason for the lack of depth being prolonged soil erosion in the past. Hughes (1989) reported that the reserve is dominated by soils of depth 300 mm or less with soil types Mispah and shallow Shortlands found extensively. This source also reported that the western part of the reserve is occupied by duplex soils, which are known for their erodibility by water owing to their textural difference between the topsoil and subsoil. A visual observation of the area also indicates that several visited areas are rocky or have considerable rocks of varying sizes.

3.5 TOPOGRAPHY

The reserve has undulating topography with wide variation in an altitude ranging between 900 and 1311 m amsl. A major division of the reserve to the north and south is created by a large valley through which the Bushman's River flows from east to west. Because of its size, this valley has made connection between the two parts difficult forcing long drives that take roads outside of the reserve to get to the other side. On either side of this valley the steepness increases drastically to reach the high grounds topped by plateaus. To the south of the river the landscape picks up height from about 900 m amsl along the Bushman's River to reach at the plateau-topped Rockmount summit with an altitude of 1311 m amsl (Hughes, 1989).

In general the reserve comprises of flat lands on the lower altitudes mainly in the western part and on plateaus both to the north and south of the Bushman's valley and steep areas that are covered by rocks along the steepness of the valley and the eastern escarpment that stretches north-south of the reserve.

3.6 VEGETATION

The vegetation of WGR can be described as *Acacia karoo-Acacia nilotica* thornveld (Edwards, 1967). The grass species of the area is highly degraded and is being invaded by scrubs and *Acacia* species (Camp, 1995a).

The reserve hosts three bioresource groups (BRG) of Camp (1999), which are given in Table 3.2 with their Acocks (1988) equivalents and their spatial distribution in the reserve is shown in Figure 3.2. Some of the most commonly found species in these groups are given in Table 3.3 (Camp, 1995a; 1995b). Predominantly, three plant species are found in the area: semi-deciduous bush, *Euphorbia* thicket and *Acacia karoo-Acacia nilotica* thornveld (Camp, 1995b). Recent study on the woody species of the reserve (Breebaart *et al*, 2001) showed the dependence of vegetation type on soil type, topographic aspect, and the presence and absence of cultivation in the past. This study has also predicted the encroachment of broadleaved plants on to many grasslands areas.

Table 3.2 *Correlation of bioresource groups with* Acocks (1975) (Camp, 1995)

Bioresource Group (Number – Name)	**Acocks (1975)** (Number – Name)
13 – Dry Tall Grassveld	65 – Southern Tall Grassveld
	64 – Northern Tall Grassveld
18 – Mixed Thornveld	65 – Southern Tall Grassveld
21 – Valley Bushveld	23 – Valley Bushveld

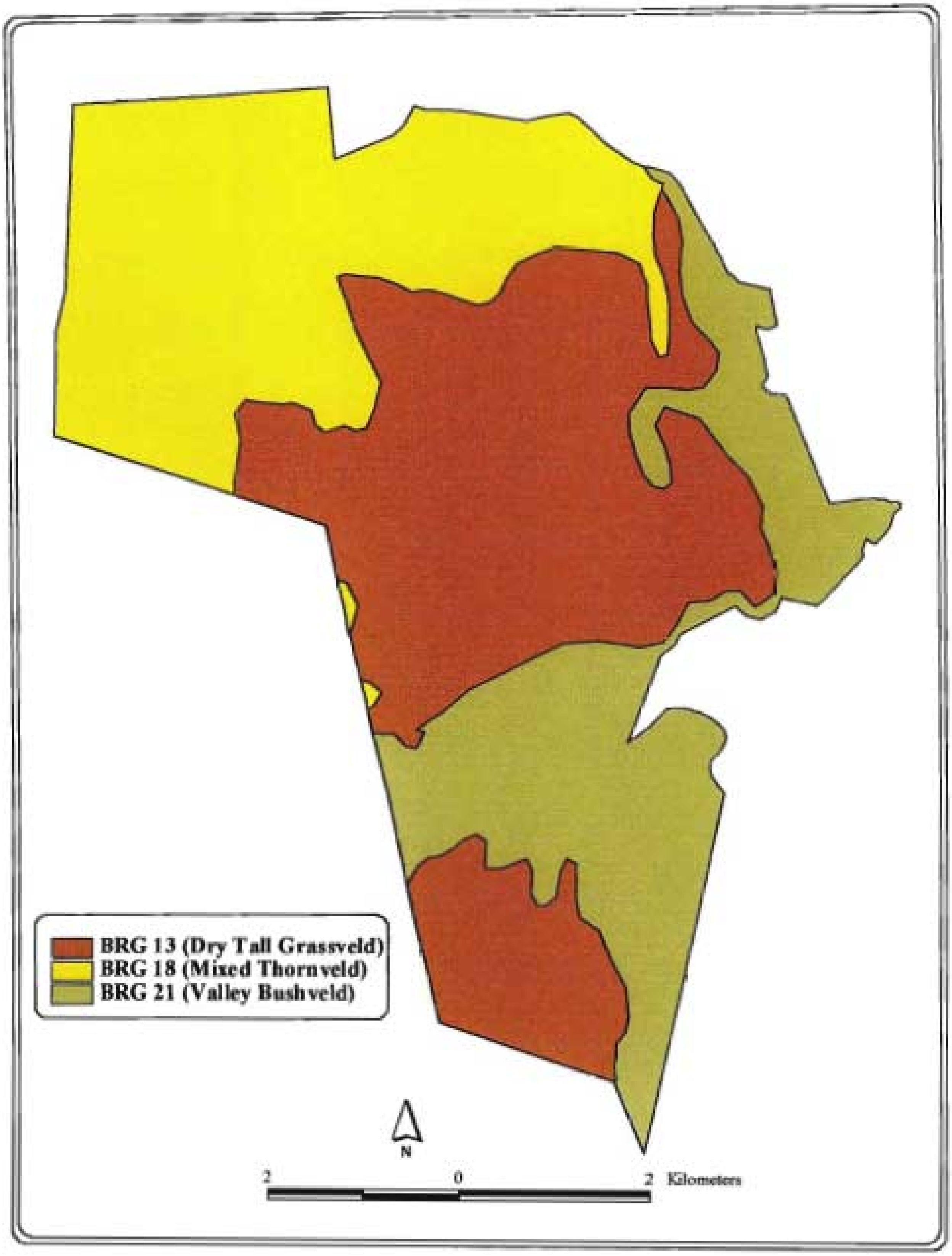

Figure 3.2 *Bioresource groups of WGR* (Camp, 1999)

Table 3.3 *Common species within the BRGs* (Camp, 1995a; 1995b)

	BRG 13	BRG 18	BRG 21
Trees and shrubs	*Acacia karroo*	*Acacia karroo*	*Acacia karroo*
	Acacia nilotica	*Acacia nilotica*	*Acacia nilotica*
	Acacia sieberana	*Acacia sieberana*	*Acacia robusta*
	Ziziphus mucronata	*Dichrostachys cinerea*	*Boscia albitrunca*
	Diospyros lycioides	*Eragrostis superba*	*Euclea crispa*
			Euclea racemosa
Grasses	*Themeda triandra*	*Themeda triandra*	*Themeda triandra*
	Hyparrhenia hirta	*Hyparrhenia hirta*	*Bothriochloa insculpta*
	Sporobolus pyramidalis	*Sporobolus pyramidalis*	*Eragrostis superba*
	Eragrostis superba	*Eragrostis superba*	*Hyparrhenia hirta*
	Aristida congesta		*Panicum maximum*
	Bothriochloa insculpta		*Sporobolus pyramidalis*
			Aristida congesta

CHAPTER FOUR: MATERIALS AND METHODS

4.1 INTRODUCTION

Soil erosion models, process-based or empirical, require the quantification of the most important factors. The RUSLE model uses five factors, namely: erosivity, erodibility, slope length and slope steepness, cover management, and conservation practice. Because of their spatial nature, the collection and analysis of such variables has always been a difficult task. However, such problem has been minimized since the development of remote sensing and GIS technologies and their involvement in hydrological models. While part of the data sets such as rainfall is acquired using direct ground measurements and others with the support of remotely sensed data as in the case of soil and land cover mapping, it has become possible to obtain data on others, for example topographic data, completely using GIS. Once the available and required data is gathered, storing and manipulating them, and producing an output for the easiest interpretation is performed in a GIS environment. This chapter discusses how the overall methodology was executed in this work. The first section deals with the data sets that were used in the study whereas the second describes the procedures undertaken to achieve the outlined objectives of the study.

4.2 DATA SETS

4.2.1 Rainfall

Based on location of rain gauge stations two sets of data were obtained in this study: one from areas outside of WGR and the other from a station found within the reserve.

4.2.1.1 Surrounding areas data set

Fifteen stations, with none in the study area, that are found within 50 km radius to the west and northwest of the reserve were selected for their proximity with their record length shown in Table 4.1. Despite the recommendation of Wischmeier and Smith (1978) and Renard *et al* (1997) to account for more than 20 years of record, the majority of them have more than 10 years long record, which might be closer to the 15-years record length requirement of Lynch and Dent (1990) to the area that includes WGR. These stations, which have autographic rainfall records with the corresponding EI_{30} values, were made available to this study by the School of Bioresources Engineering and Environmental Hydrology (BEEH) at the University of KwaZulu-Natal.

Table 4.1 *Locations, place names, and record lengths of surrounding areas stations*

	Location		Record Length		
Station ID	**Latitude**	**Longitude**	**Period**	**Length (yrs)**	**Place name**
1	29.6653	-29.0019	1977-88	11	DE HOEK
4	29.6528	-29.0111	1964-69; 1980-88	19	DE HOEK
11	29.6272	-29.0122	1976-94	19	NTABAMHLOPE
14	29.6658	-29.0344	1970-72; 1976-96	24	NTABAMHLOPE
18	29.6619	-29.0406	1976-96	21	NTABAMHLOPE
20	29.6725	-29.0194	1985-95	11	NTABAMHLOPE
21	29.6464	-29.0442	1990-95	6	NTABAMHLOPE
23	29.6564	-29.0581	1964-96; no 1986	32	NTABAMHLOPE
40	29.5983	-29.0356	1988-96	9	DRIEFONTEIN
41	29.6289	-29.0683	1990-96	7	CRISTALSPRS
0268631	29.8667	-29.0167	1959-73	15	ESTCOURT
0300423	29.7500	-28.5500	1967-79	13	LADYSMITH
0300454	29.7667	-28.5667	1949-55; 1979-90	19	LADYSMITH
0300483	29.7833	-28.5500	1956-65	10	LADYSMITH
0300690	29.8833	-29.0000	1973-96	24	ESTCOURT

4.2.1.2 Weenen Game Reserve data set

The reserve has only one rain gauge station from which monthly rainfall records gathered by WGR staff that run from 1981 through 2002 were taken. The 22 years of record is a typical length to deriving long-term annual erosivity factor (Wischmeier and Smith, 1978; Renard *et al*, 1997). After removing the no-rain months, 233 months were taken. The records of these months were, then, run using the developed equation and the results were added within their respective years. Finally, the annual values were averaged over the 22 years.

4.2.2 Soil

Soil erodibility as used in the RUSLE is a function of soil texture, soil structure, and soil organic matter content (Wischmeier and Smith, 1978; Renard *et al*, 1997). In order to determine these characteristics 53 soil samples were taken. Since soil erosion is a surface phenomenon, samples were taken from the top 15 cm of the soil surface as proposed by the RUSLE guideline (Renard *et al*, 1997). In the sampling, the soil map of the area at a scale of 1:10 000 (Hughes, 1989) was used and the representation of the soil types by the sample size and the area covered by each type were balanced (Figure 4.1). The major soil type in the reserve is Mispah (Ms) and the number of samples taken from this soil type is the largest. Forty-six soil samples were taken from the area north of the Bushman's River, as this was the larger area. See appendix for brief description of the soil forms in WGR.

Instruments that were used to collect these soil samples in the field were auger and hoe, plastic bags, GPS instrument. Laboratory equipments used to make textural analysis of these samples were mortar and pestle, sieves, beakers, Labsonic 1510 ultrasonic probe, measuring cylinders, weighing dishes, Bel electronic balance model Mark 500, plunger, stopwatch, oven, and hydrometer, whereas the reagents used were sodium carbonate, sodium hexametaphosphate, and de-ionized water. For organic carbon analysis the following materials were used: 0.5 mm sieve, 50 ml glass burette, and Erlenmeyer flask and the reagents were 1N potassium dichromate, 0.5N ferrous ammonium sulphate, FERROIN indicator, sodium fluoride (NaF), phosphoric acid (H_3PO_4), concentrated sulphuric acid (H_2SO_4).

4.2.3 Topographic parameters

The parameter required to obtain the topographic factor is a Digital Elevation Model (DEM) of the study area. This was provided in grid format for the whole of KwaZulu-Natal by the Chief Directorate: Surveys and Mapping at Cape Town from which the data for the study area was extracted. The original data was in WGS and hence had to be projected to Transverse Mercator, Clarke 1880, and central meridian of 31° longitude. The grid cell size was a square of about 22.7 m at each side and this was resampled to 30 m to conform to the grid cell size of all the other factors.

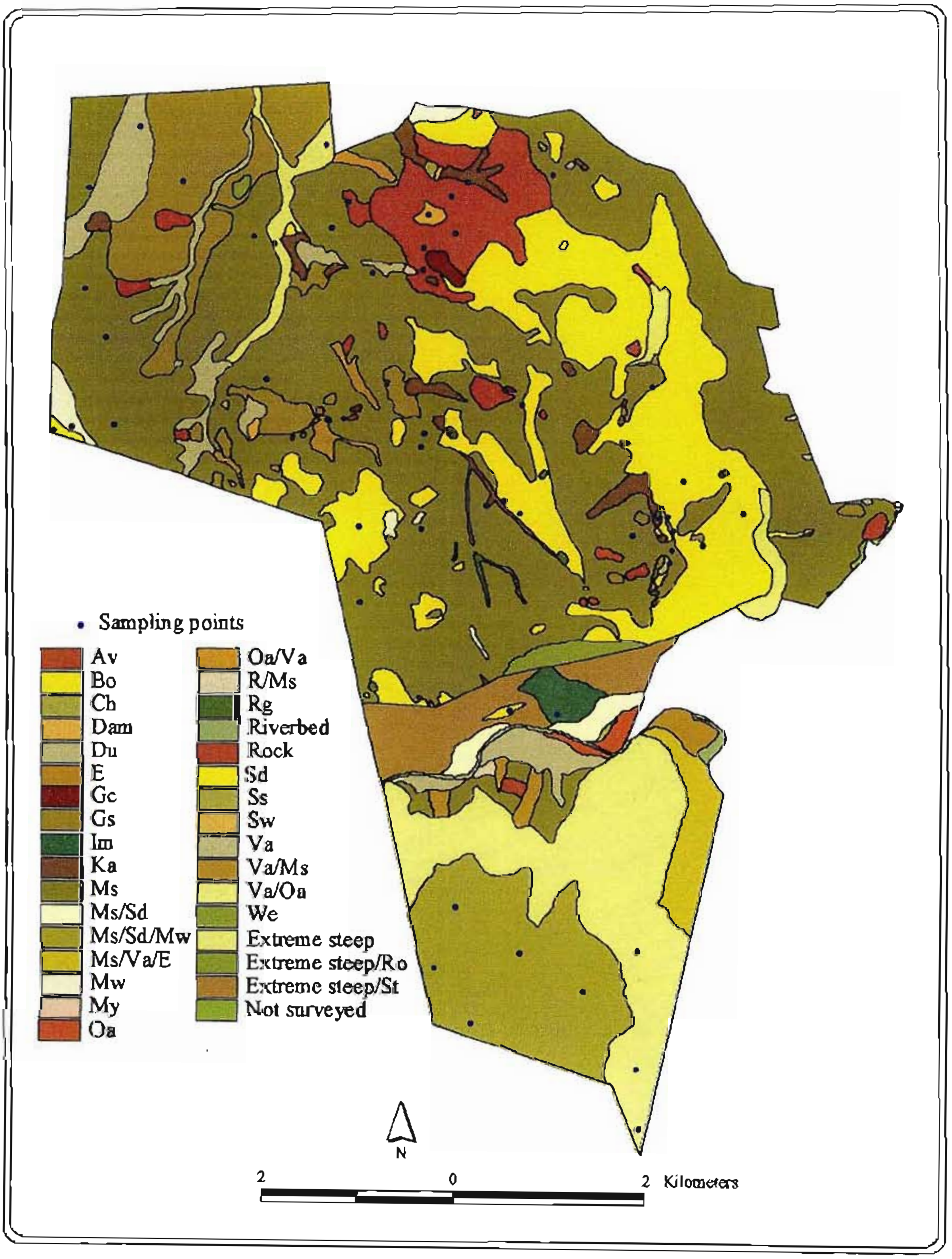

Figure 4.1 *WGR Soil types (*Hughes, 1989*) and sampling points' map*

4.2.4 Vegetation cover

Data to be used in the USLE C factor table for rangelands are canopy height, canopy cover, and mulch cover, which also requires for the consideration of the mulch type whether it is dominated by grass and/or grass like species or broad leaf plants at least 5 cm deep (Wischmeier and Smith, 1978). The canopy height is given in meters and is applied mainly to trees as they have canopy heights compared to grasses. Canopy cover is the area of land that is protected by the canopy size and is given in %. As with canopy height, the need to determine canopy cover is applicable more to trees. Mulch cover was the parameter that was best quantified in this study using a stick. The USLE C factor table (Table 5.12) was also used.

Also, a satellite image of Landsat 7 Enhanced Thematic Mapper plus (ETM+), which was taken in March 2002 was used. This image was selected since it was, after visual examination, found out to be representative of the vegetation condition of the area, which attains its optimum vigorous stage during the aforementioned time following Tueller's (1991) recommendation to select remotely sensed images based on season where the maximum separation of vegetation from bare ground can be obtained.

4.3 METHODS

4.3.1 Rainfall analysis

The rainfall erosivity factor accounts for the effect of raindrops and surface runoff, and assumes that, if all other factors are held constant, the amount and strength of rainfall is directly related to soil loss (Wischmeier and Smith, 1978). The RUSLE just like its predecessor, the USLE, uses EI_{30} index, that is, the product of storm energy and maximum rainfall amount falling within 30 consecutive minutes. In calculating this factor a storm is defined which is expressed as an amount with a minimum of 12.5 mm within 6 consecutive hours. However, this definition also allows the consideration of storms with less than 12.5 mm provided that 6.35 mm or more fall within 15 minutes (Wischmeier and Smith, 1978; Renard *et al*, 1997). Consequently, calculating EI_{30} requires temporally detailed rainfall records obtainable from autographic recording stations, which are not available in most cases owing to their costliness. Rather, equivalent equations based on limited autographic data are developed to use more easily available records such as daily rainfall data (Nwosu *et al*, 1995; Yu, 1998; Yu and Rosewell, 1996a; Yu *et al*, 2001;

Loureiro and Coutinho, 2001), monthly rainfall data as reviewed by Renard and Freimund (1994), and annual precipitation (Yu and Rosewell, 1996b; Millward and Mersey, 1999).

Correlation of EI_{30} with easily available data was proposed by Smithen (1981) and Smithen and Schulze (1982) for South Africa who used 4 different approaches. These are the total rainfall, the effective rainfall, the Modified Fournier's Index, and Burst Factor, which all use monthly rainfall data. From the results, the Cedara station, which might have represented the present KwaZulu-Natal Province, is best explained by the Burst Factor (r=0.97). This factor is given as:

$$\mathrm{BF} = \sum_{i=1}^{12} \frac{M_i Pe_i}{P} \tag{4.1}$$

where BF is Burst Factor, M_i is maximum daily rainfall for month i in mm, Pe_i is effective rainfall in mm for month i and P is annual rainfall in mm. Effective rainfall includes all storms except those less than 12.5 mm separated by more than 6 hours. This definition of rainfall, therefore, requires rainfall records against shorter time interval than 24-hours period, which still may pose a problem to daily recording stations. Moreover, this approach covers South Africa at a small scale failing to represent small areas adequately. Therefore, developing relations using the data discussed in Section 4.2.1 were performed as described in the next paragraph.

The data sets of the surrounding areas and WGR were first compared using basic statistical descriptions. Then, the surrounding areas data set, which has EI_{30} values derived using the RUSLE guideline, was dealt with initially. Before further analysis the EI_{30} values of the storms of the surrounding areas were divided by 1000 for simplifying the computation as recommended to South Africa by Smithen (1981) and Smithen and Schulze (1982). Following this, rainfall depths of daily, monthly, and yearly were related with the respective EI_{30} values of the surrounding areas. Multiple linear regression analysis using quadratic function was applied to develop each equation that relates amount of rainfall depths and EI_{30} values. The resulting equations were then compared by their coefficients of determination and correlation coefficients of the observed and the estimated values. The best relation with the highest values of these comparisons was then selected to be applied for the data set of WGR.

4.3.2 Soil erodibility

The laboratory texture analysis of the soil samples was undertaken using the hydrometer method (Gee and Bauder, 1986; Sheldrick and Wang, 1993), which is briefly summarized as follows. According to the model, soil samples were air-dried and ground to separate soil samples having particle size of less than 2 mm from each. After mixing these samples with distilled water, they were dispersed with ultrasonic probe and then transferred into a 1 litre-measuring cylinder with distilled water. Then, using a hydrometer, which measures the density of the solution, readings were taken at 2 different times the earlier being for sand and then for clay, which respectively need less and more time for settling in the solution. The silt content was then inferred from these measurements. The numerical values obtained as a result of this analysis were assigned textural names, in turn, to be refined according to the requirements of the RUSLE guideline (Table 4.2).

Table 4.2 *Textural assignments of the RUSLE* (Renard *et al*, 1997)

Texture class	Size (mm)
Sand	0.1-2.0
Very fine sand	0.05-0.1
Silt	0.002-0.05
Clay	<0.002

The soil texture classes again were used to assign soil permeability values of the RUSLE guideline (Table 4.3). These permeability values were given based on the contribution of the soil texture class towards generating surface runoff. Thus, soils with low infiltration and permeability capacity have high runoff potential and hence are given higher permeability value with maximum of 6 and soils with high infiltration and permeability capability have low surface runoff, which are given low permeability values that could reach to the minimum of 1.

The organic matter content of the samples was determined using Walkley-Black method (Nelson and Sommers, 1996). In this method each air-dry soil sample was ground and passed through 0.5 mm sieve, which was then mixed with potassium dichromate and sulphuric acid. After cooling, the following reagents were added: distilled water, phosphoric acid, and solid sodium fluoride. This mixture was then titrated.

Soil structure is the aggregate stability of the soil. Its determination was made by visual observation in the field and hand crushing soil mass to get the pedons for assigning the structure codes as given in (Table 4.4).

Table 4.3 *Permeability values for major textural classes* (Renard *et al*, 1997)

Texture Class	Permeability Class, P_s	Saturated Hydraulic Conductivity (mm hr^{-1})
Clay, Silty Clay	6	<1
Silty Clay Loam, Sandy Clay	5	1 – 2
Sand Clay Loam, Clay Loam	4	2 – 5
Loam, Silty Loam	3	5 – 20
Loamy Sand, Sandy Loam	2	20 – 60
Sand	1	>60

Table 4.4 *Soil structure class and their values* (Wischmeier and Smith, 1978)

Soil structure	Structure Class, S_s
Very fine granular	1
Fine granular	2
Medium or coarse granular	3
Blocky, platy or massive	4

The above values were then run into the soil erodibility nomograph equation of the RUSLE model (Equation 4.2, adapted by Lorentz and Schulze (1994)) with results representing point erodibility (K) values at the respective sampling locations.

$$K = 0.01317\,[0.00021(12 - OM)\,M^{1.14} + 3.25(S_s - 2) + 2.5(P_s - 3)] \tag{4.2}$$

where,

OM is organic matter content given in %

M = (% modified silt + % silt) × (% modified silt + % sand)

S_s is the structure code

P_s is the permeability class

Since these values represent only sampling points, the non-sampled areas had to be filled in by interpolating the calculated point values in ArcView GIS. For the interpolation, 38 randomly chosen points based on their spatial locations were taken and the remaining 15 were used to assess the accuracy of the interpolation (Ahmed, 2003). The three interpolation methods, namely, spline, inverse distance weighting (IDW), and ordinary kriging were tested for the best fit.

4.3.3 Topographic factor

In any hydrological modeling that uses DEM, the primary requirement is assurance of depressionless elevation surface, which are usually considered as errors (ESRI, 1996; Hickey, 2000). DEMs with depressions/sinks obstruct flow that creates discontinuity of the latter and a grid or a set of grids with lower altitude than the surrounding cells is the cause for this. Removal of depressions, which works based on increasing the altitude of the sink to a level of the lowest neighbouring cell, was applied to the DEM of WGR within ArcView GIS.

Topographic factor, LS, is soil loss ratio from a land to that from a standard plot of 22.1 m long and 9% slope (Wischmeier and Smith, 1978; Renard *et al*, 1997). This factor combines the slope length and the gradient of a given segment of land. In a uniform slope condition, Equation 4.3 is applied.

$$LS=(\lambda/22.1)^{m}\,(0.065+0.045\sin s+0.0065\sin s^{2}) \tag{4.3}$$

where LS is the combined length-slope factor; λ is slope length in meter; s is gradient in %; and m is length exponent specified as 0.5 for slopes of 5% or more, 0.4 for slopes between 3.5 and

4.5%, 0.3 if slope is between 1 to 3%, and 0.2 if slope is less than 1% (Wischmeier and Smith, 1978).

In a complex topographic condition the RUSLE applies Equation 4.4 of Foster and Wischmeier (1974).

$$LS = \frac{S_j(\lambda_j^{m+1} - \lambda_{j-1}^{m+1})}{(\lambda_j - \lambda_{j-1})(22.13)} \quad (4.4)$$

where L is slope length factor for the j-th segment, Sj is slope factor for the j-th segment, λj is distance from the lower boundary of the j-th segment to the upslope field boundary, and m is the length exponent of the USLE LS factor. Later, Desmet and Govers (1996) modified this formula as Equations 4.5 a and b:

$$L_{i,j} = \frac{A_{i,j-out,}^{m+1} + A_{i,j-in}^{m+1}}{(A_{i,j-out} - A_{i,j-in})(22.13)^m} \quad (4.5a)$$

where $L_{i,j}$ is the slope length factor of the cell with coordinates (i, j), $A_{i,j\text{-}out}$ in m^2 is the contributing area at the outlet of the grid cell with coordinates (i, j), $A_{i,j\text{-}in}$ in m^2 m^{-1} is the contributing area at the inlet of the grid cell with coordinates (i, j), and m is the slope length exponent of the RUSLE S-factor.

$$S = -1.5 + \frac{17}{(1 + e^{2.3-6.1\sin\theta})} \quad (4.5b)$$

where θ in degrees is the slope angle of the grid cell.

The advantage of this modification is that it works on the basis of flow accumulation. Mitasova *et al* (1996) further simplified the formula to derive the LS factor at a point and produced the following GIS executable equation (Equation 4.6):

$$LS(\mathbf{r}) = (m+1)\ [A(\mathbf{r})/a_0]^m\ [\sin b(\mathbf{r})/b_0]^n \qquad (4.6)$$

where LS is the combined length-slope factor at a given point **r** with coordinates x and y. A is the unit contributing area per unit contour width in meter, which is perpendicular to the flow direction (aspect); b is the slope in degree; a_0 is 22.1 m, the slope length of standard USLE plot; and b_0 is 0.09 (9%), slope gradient of the standard USLE gradient. m and n are parameters set as 0.6 and 1.3, respectively. These constant values are taken because their results conform with slope length factor if slope length is less than 100 m and gradient is less than 14 degree (about 25%) (Moore and Wilson, 1992).

4.3.4 Cover management factor

The cover management factor is considered as the most complex and demands caution in its calculation. One of the major improvements in the accuracy of this factor within the RUSLE (Renard *et al*, 1997) model is the use of sub-factors, which are discussed in Section 2.6.3.4.4. However, in this study, undertaking the measurements of these sub-factors in their entirety as outlined in the RUSLE guide was seen as costly and time prohibitive. These problems obliged this study to follow a simpler approach with the awareness of loss in accuracy of the result. For permanent rangeland scenario, Lorentz and Schulze (1994) recommend assigning values for this factor from the USLE table once the cover density is quantified. Yet, unlike croplands whose uniformity in species composition and growth rate simplify deriving parameters from them, rangelands show spatial and temporal complexity in their vegetation characteristics that makes their quantification difficult (Abel and Stocking, 1987; Bartsch *et al*, 2002). Moreover, the USLE rangeland C values of Wischmeier and Smith (1978) were never derived from nor validated on rangelands (Foster, 1982).

To determine these parameters about 30 m transect lines were taken based on the image of Landsat 7 ETM+ to get % ground cover using point-intercept method (Floyd and Anderson, 1987; Diersing *et al*, 1992, cited in Wang *et al* (2002)). Perpendicular points were taken using a stick at 1 m interval making a total of 30 points at each transect line. Where the topographic conditions did not allow applying this, visual estimations were made. Sykes *et al* (1983) concluded that results obtained by using this technique are quick with yet acceptable accuracy.

Points that are covered with vegetation and rocks, of which the latter are assumed to have similar effect as mulch cover if existing on the surface (Renard *et al,* 1997), were then divided by the total number of points along each transect and the results were multiplied by 100 to get % value. The resulting cover estimates were then used to assign values from the USLE C factor values table (Wischmeier and Smith, 1978), which is also readily adopted by Lorentz and Schulze (1994) within the ACRU hydrological modeling program for permanent rangeland scenario.

Conversion of point values using spatial interpolation is far from expressing the ground reality adequately (Wang *et al*, 2002). Though at its infant stage, using remote sensing technology in estimating cover factors by correlating it with ground collected value is promising. Such approaches were employed by various researchers such as De Jong (1994a; 1994b), Ehlschlaeger and Tweddale (2000), and Wang *et al* (2002). The main principle of these studies was the use of various vegetation indices from which pixel values were read to correlate with ground truth points. The most widely used vegetation indices and their equations, which were also applied in this study, are:

$$\mathrm{NDVI}=\frac{NIR-\mathrm{Re}\,d}{NIR+\mathrm{Re}\,d} \tag{4.7}$$

$$\mathrm{TNDVI}=\sqrt{\frac{NIR-\mathrm{Re}\,d}{NIR+\mathrm{Re}\,d}+0.5} \tag{4.8}$$

$$\mathrm{SAVI}=\frac{NIR-\mathrm{Re}\,d}{NIR+\mathrm{Re}\,d+L}(1+L) \tag{4.9}$$

$$\mathrm{TSAVI}=\frac{a(NIR-a\,\mathrm{Re}\,d-b)}{\mathrm{Re}\,d+aNIR-ab} \tag{4.10}$$

$$\mathrm{MSAVI}=\frac{2NIR+1-\sqrt{(2NIR+1)^2-8(NIR-\mathrm{Re}\,d)}}{2} \tag{4.11}$$

where *NIR* is Near Infra Red; NDVI is Normalized Difference Vegetation Index (Tucker, 1979); TNDVI is Transformed Normalized Difference Vegetation Index (Huete *et al*, 1985; Huete and Jackson, 1987); SAVI is Soil Adjusted Vegetation Index (Huete, 1988); TSAVI is Transformed Soil Adjusted Vegetation Index (Baret, 1989, cited in Purevdorj *et al* (1998)); and MSAVI is Modified Soil Adjusted Vegetation Index (Qi *et al*, 1994). L is an adjustment factor usually specified to be 0.5, which is recommended for intermediate vegetation cover condition. The TSAVI considers L as self-adjusting factor that depends on the slope, a, and intercept, b, of a soil line curve. This line was produced by plotting pixel values, which are considered to represent bare soils, on a two dimensional axes of Red (TM band 3) as the x axis and NIR (TM band 4) as the y axis.

Centering each ground point, where the C values are known, 5 pixels by 5 pixels (150×150 m) were considered to get an average value representing an area of 2.25 ha from each vegetation-index-transformed image. These pixel values served as independent variables against the assigned USLE C values as dependent variables to derive an equation for each vegetation index considered using multiple linear regressions. Comparisons of the different vegetation-indices-derived equations were then made to select the one that correlates best with the ground information. These include coefficients of determination, standard error of prediction, and correlation coefficients between the estimated and the calculated C values.

4.3.5 Soil loss

The RUSLE calculates soil erosion by multiplying the factors listed in the previous subsections. Application of this equation in GIS follows the conversion of conformity of all these factors regarding grid cell size. After specifying a side of 30 m grid cells for each factor, all the four factors were multiplied with results to be presented per grid cell. The results were then interpreted based on their severity and comparisons were made with related works done elsewhere.

4.4 SUMMARY

Relevant data that are required by the RUSLE soil erosion model were gathered. Soil data and rainfall records were taken from the field whereas topographic data was obtained in the form of gridded Digital Elevation Model (DEM). Satellite imagery of Landsat 7 ETM+ was employed for deriving the cover management factor. These data were then analyzed as outlined in the form of a flow diagram shown in Figure 4.2. The conservation practice factor (P) is not included in the diagram since it was taken to be 1 throughout the study area and has no effect in the final result of the multiplication.

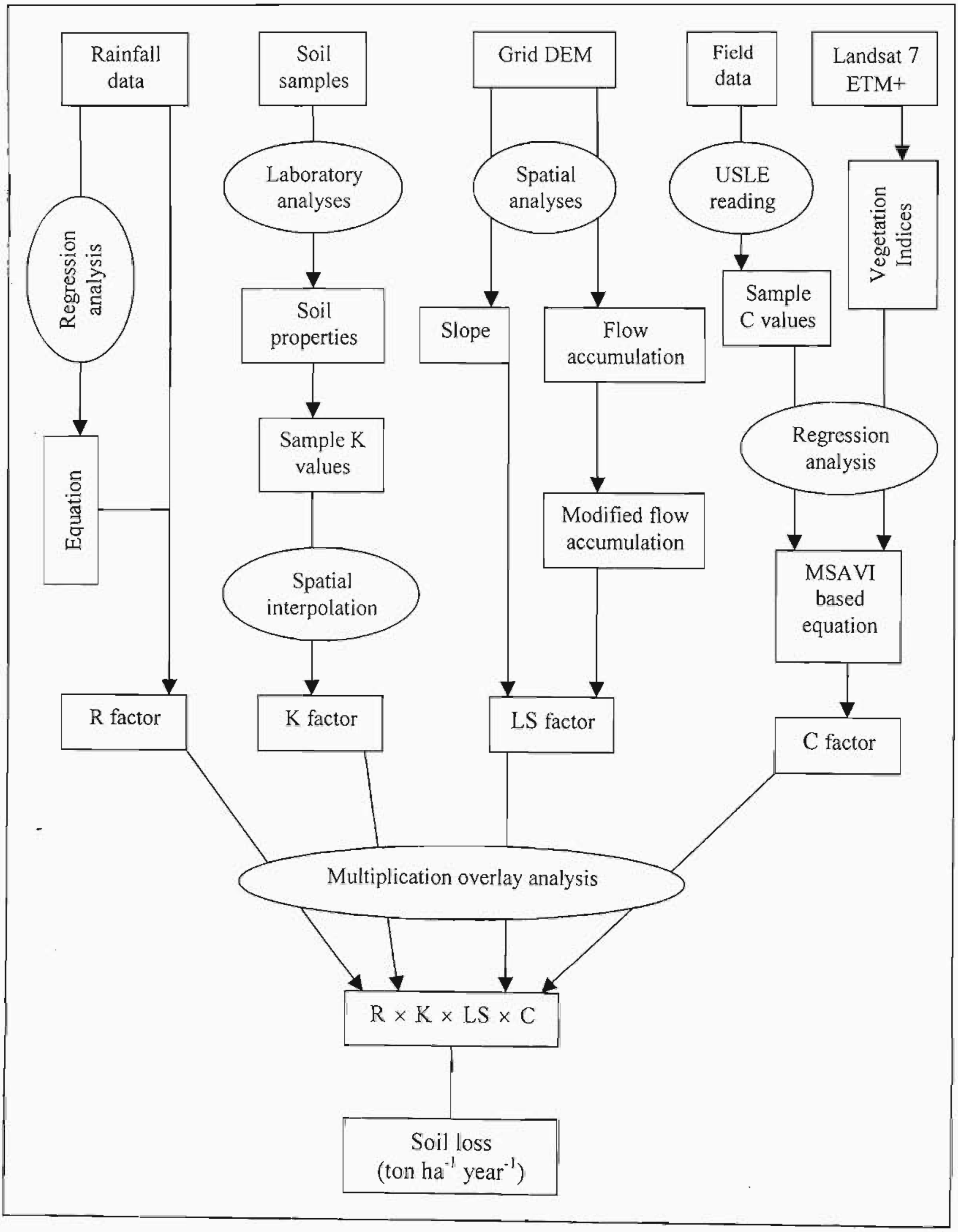

Figure 4.2 *General flow diagram summarizing the analysis methods used in the study*

CHAPTER FIVE: RESULTS AND DISCUSSION

In this chapter results of the analyses of the data sets using the methods outlined in the previous chapter are presented. The results include laboratory, statistical, GIS, and remote sensing analyses. Graphical and diagrammatic representations accompany the results. This is followed by the discussions of the results. Whenever feasible and appropriate related case studies conducted elsewhere are brought in for comparison.

5.1 RESULTS

The results of this study are presented according to the methods followed. The factors generated are presented first and the final result combining all the factors is presented thereafter.

5.1.1 Rainfall erosivity factor (R)

5.1.1.1 Comparison of the surrounding areas (15 stations) and WGR data sets

The comparison of the data sets for the surrounding areas and WGR using basic statistical descriptions is given in Table 5.1 and Figure 5.1.

The mean monthly depth for the surrounding areas, 69 mm, is much lower than the maximum value, 407 mm, that can be ascribed to the fact that only 25% (upper quartile) of the total number of months has rainfall depth of 104 mm or more (Table 5.1). Also, the monthly rainfall depth shows high variation among the months with coefficient of variation (CV) of 94%. Looking at the seasonality of the data, the summer months, October to March, have the higher share, about 83.8% of the annual figure (Table 5.1 and Figure 5.1). The CV of the summer months and April is lower whereas for the rest of the year is higher than the total monthly value (Figure 5.1).

Similar to the surrounding areas data, the mean value of WGR, 68 mm, is much less than the maximum, 322 mm, since only 25% (upper quartile) is more than or equal to 109 mm (Table 5.1). The CV for the WGR data set is about 91%, which is comparable to the surrounding areas stations. Studying the seasonal distribution, this data set displays similar characteristics as records of the surrounding stations in that the proportion of the summer season from the annual is high, 84.1% and the CV during this time is lower than the remaining (Figure 5.1).

Table 5.1 *Mean monthly rain depth comparisons of the surrounding areas and WGR*

Monthly depth	Min	Max	Mean	CV (%)	Lower Quartile	Upper Quartile	Summer share (%)
Surrounding stations	0.1	407	69	94	16	104	83.8
WGR station	1	322	68	91	18	109	84.1

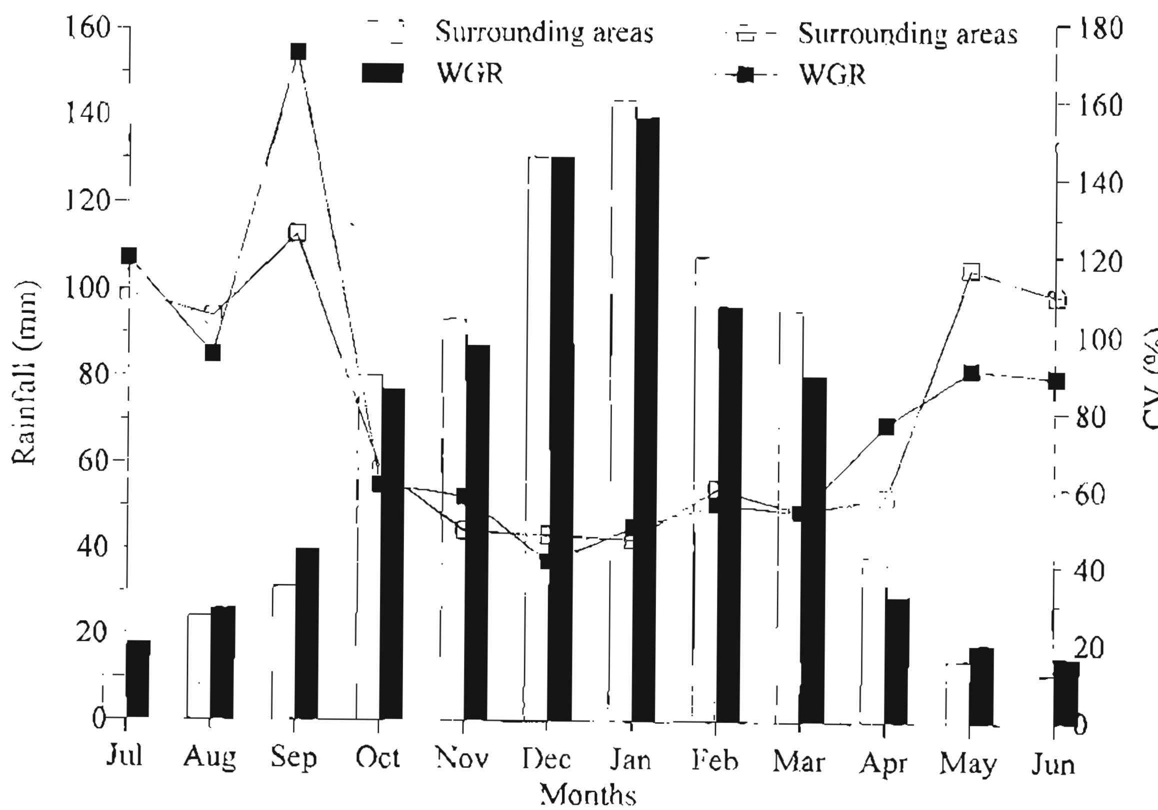

Figure 5.1 *Monthly mean depth and CV (%) comparisons of the surrounding areas and WGR*

5.1.1.2 Deriving erosivity equation

Results of the various parameters against the EI_{30} values are given in Table 5.2. The best parameter was obtained using the monthly rainfall and the equation (Equation 5.1) developed from this parameter has the highest coefficient of determination, r^2, of 0.733 (Table 5.2), implying that 73.3% of the variance in erosivity (R) can be explained by total monthly rainfall (P) at $p<0.001$.

$$R = -1.675+0.1489\,P+0.001757\,P^2 \quad (r^2=0.733) \tag{5.1}$$

where,

R = Erosivity (MJ mm ha^{-1} hr^{-1}), and

P = Total monthly rainfall (mm)

The graph of the estimated values versus the original values (Figure 5.2), which has the highest correlation coefficient, r, of 0.86 (Table 5.2), indicates a fairly comparable relation with a fitted line having a slope of 0.733 as compared to what would have been 1 for a perfect match. Meanwhile, it is noteworthy to mention the existence of negative values in the estimated data set. Since these values are results of the developed equation, they also occur when running the data of the study area. Assuming a minimum possible measuring value, a zero was substituted for every negative number. On the other hand, excluding the negative values from the record would imply that there was no rain in the particular month.

Table 5.2 *Comparison of various rainfall depth parameters*

Parameter	Coefficient of determination (r^2)	Correlation coefficient (r)
Daily	0.640	0.80
Monthly	0.733	0.86
Yearly	0.541	0.74

5.1.1.3 Weenen Game Reserve EI_{30}

Using Equation 5.1, the monthly precipitation records of the study area were calculated and the results are averaged by months. Basic characteristics of the results along with comparisons with the surrounding areas data set are given in Table 5.3 and Figure 5.3. The mean monthly EI_{30} value is 24, which is equal to the surrounding areas data set and the CV(%) values are considerably high for both data sets. Analysis on monthly basis shows that summer (October to March) has a far greater share of the mean monthly value for both data sets with comparable magnitudes.

After adding the monthly values within the respective years, average annual value was calculated as 249 with standard deviation of 91. The deviation of each year from the mean is illustrated as in Figure 5.4, which clearly indicates some years having much higher erosivity magnitudes underlying the reason for considering a long time period. On the basis of the RUSLE guideline, the average value over the years was taken and this represented the whole study area though it might not be the case in reality; because there is a general belief that the amount of precipitation in the study area varies with topography (Breebaart *et al*, 2001). However, analysis of monthly rainfall of 101 stations and their altitude within 50 km radius of the study area did not show any relationship between them.

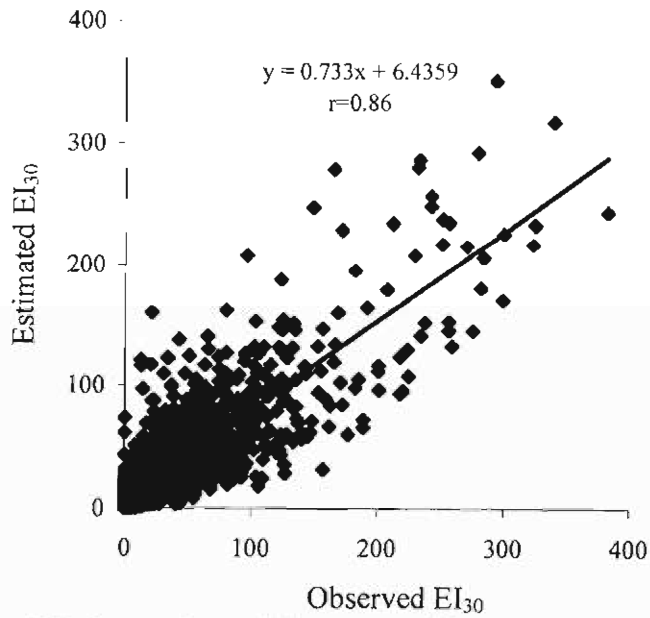

Figure 5.2 *Comparison of observed and estimated EI_{30}*

Table 5.3 *Mean monthly EI_{30} comparisons of the surrounding areas and WGR*

Monthly EI_{30}	Min	Max	Mean	CV (%)	Lower Quartile	Upper Quartile	Summer share (%)
Surrounding stations	0	350	24	177	1	33	95.9
WGR station	0	227	24	142	2	35	89.7

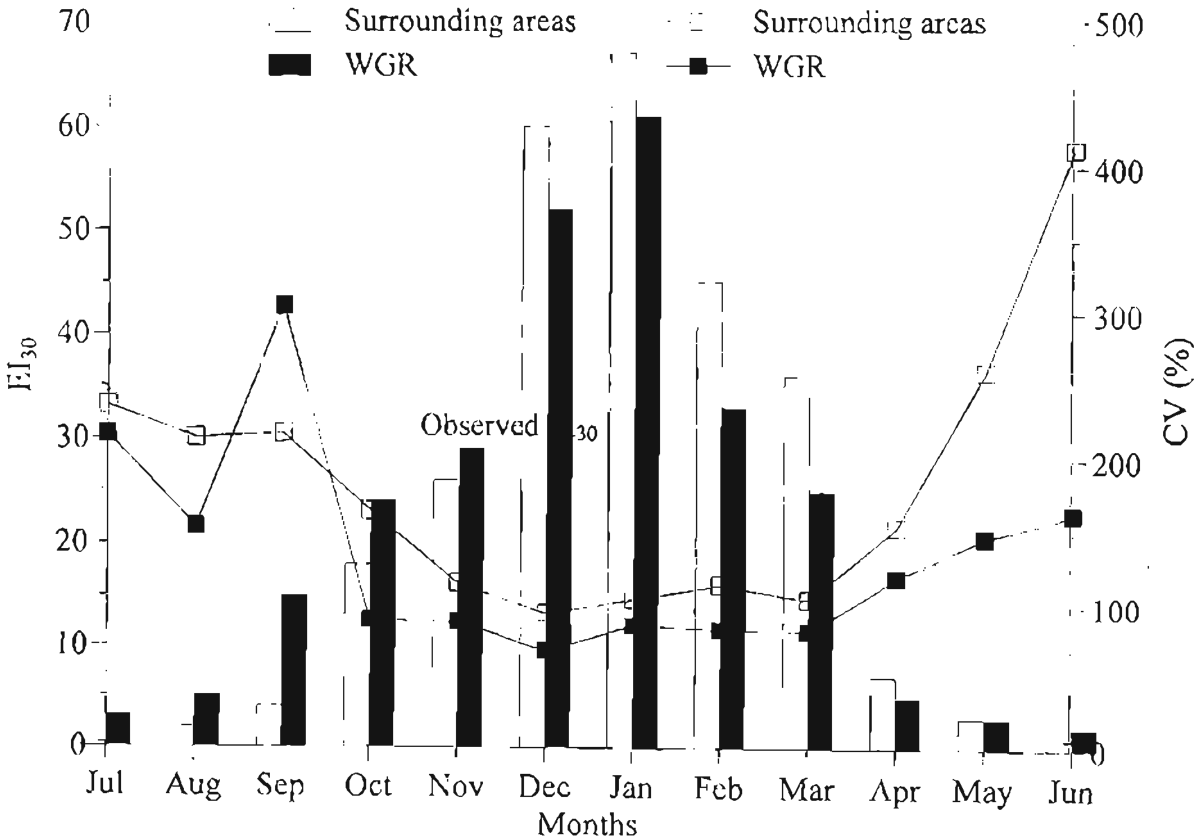

Figure 5.3 *Monthly mean EI_{30} and CV (%) comparisons of the surrounding areas and WGR*

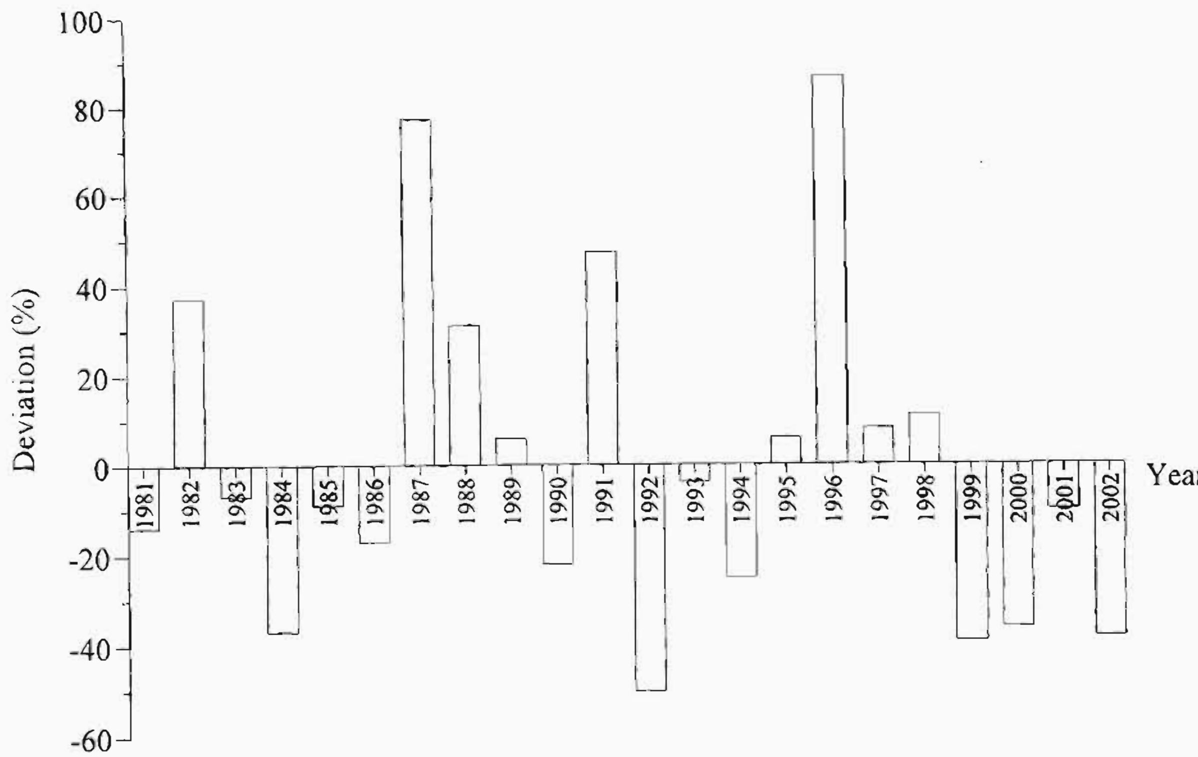

Figure 5.4 *Annual % deviation of EI_{30} from the mean for WGR*

5.1.2 Soil erodibility factor (K)

5.1.2.1 Sample point values

Results of soil sample analyses that represent the point observations are given in Table 5.4. The textural analysis shows that, 22 soil samples have clayey texture. The permeability classes (column 12), assigned based on these textural classes, indicate that 22 soil samples have a class of 6, that is, their contribution towards surface runoff is the highest, whereas only one sample has the lowest runoff-contribution potential. The average organic matter content of the samples is 2.74%. From the total number of samples, 26 have organic matter content (column 9) greater than the mean value. Out of these, 11 had more than 4%, which could not be read off from the soil erodibility nomograph due to which they were adjusted so that the maximum threshold is set to 4% (column 10, given as Adj. %OC).

The structure code, which was obtained by observing the virgin soil, shows that most of the samples have fine granular structure.

The resulting K values, obtained using Equation 4.2, are given in column 14. The minimum and maximum values are 0.090 and 0.422, respectively. The mean value is 0.210 and the coefficient of variation (CV) of the samples in K value is 35%.

Table 5.4 *Soil sampling locations and laboratory analyses results*

Sample	Latitude	Longitude	%Silt	%VFS	%Sand	%Clay	%Modified silt	%OC	Adj. %OC	Texture	Perme-ability Class	Structure Code	**Erodibility (K) value**
1	-28.86048	29.96733	14.80	9.48	36.36	39.36	24.28	0.92	0.92	SaCl	5	1	0.148
2	-28.86062	29.96281	15.68	12.47	32.49	39.36	28.15	3.11	3.11	SaCl	5	1	0.142
3	-28.86095	29.96090	17.13	10.55	18.57	53.75	27.68	3.11	3.11	Cl	6	3	0.227
4	-28.85220	29.96649	23.71	16.87	49.82	9.61	40.58	0.88	0.88	SaCl	5	2	0.422
5	-28.84779	29.96441	15.92	11.65	64.38	8.14	27.57	2.63	2.63	SaLo	2	2	0.164
6	-28.83779	29.97503	16.13	19.65	51.82	12.41	35.77	1.28	1.28	SaLo	2	2	0.254
7	-28.83265	29.97047	8.01	16.07	73.20	2.71	24.09	0.92	0.92	Sa	1	3	0.190
8	-28.83836	29.96499	10.20	12.25	36.34	41.21	22.45	1.20	1.20	SaCl	5	2	0.174
9	-28.84311	29.98258	13.60	9.53	67.08	9.80	23.12	2.67	2.67	SaLo	2	2	0.124
10	-28.84375	29.98460	15.99	16.86	50.43	16.72	32.85	2.27	2.27	SaLo	2	1	0.147
11	-28.84233	29.99075	8.29	13.96	60.49	17.25	22.26	1.04	1.04	SaLo	2	2	0.127
12	-28.83992	29.99262	12.71	12.34	24.99	49.96	25.05	3.79	3.79	Cl	6	2	0.176
13	-28.92668	30.02283	15.47	13.58	46.07	24.88	29.05	4.71	4.00	SaClLo	4	2	0.175
14	-28.92094	30.02273	12.50	14.13	43.14	30.22	26.64	3.95	3.95	SaClLo	4	2	0.152
15	-28.90987	30.02273	10.49	9.83	28.75	50.93	20.32	6.30	4.00	Cl	6	3	0.200
16	-28.91368	30.01689	25.67	10.66	14.15	49.52	36.33	2.87	2.87	Cl	6	2	0.231
17	-28.91002	30.01028	23.17	20.83	13.18	42.82	44.00	4.47	4.00	Cl	6	3	0.308
18	-28.90579	30.00336	15.66	13.48	18.27	52.59	29.14	3.23	3.23	Cl	6	1	0.148
19	-28.91130	30.00105	15.07	12.09	12.72	60.11	27.17	4.31	4.00	Cl	6	3	0.205
20	-28.91657	30.00505	11.75	28.93	27.00	32.33	40.68	3.35	3.35	SaClLo	4	2	0.233
21	-28.88773	30.01444	26.90	30.14	39.89	3.07	57.04	3.19	3.19	SaLo	2	2	0.417
22	-28.88734	30.00927	29.88	10.60	18.19	41.32	40.49	4.99	4.00	Cl	6	3	0.298
23	-28.86930	30.03430	19.37	24.74	28.81	27.07	44.11	1.68	1.68	SaClLo	4	2	0.317
24	-28.86624	30.02794	25.35	11.33	21.36	41.96	36.69	4.59	4.00	Cl	6	3	0.279
25	-28.86957	30.02612	17.48	5.16	11.50	65.86	22.64	4.43	4.00	Cl	6	3	0.185
26	-28.87084	30.02677	21.92	8.89	17.54	51.66	30.80	5.07	4.00	Cl	6	1	0.148

Table 5.4 *Continued*

Sample	Latitude	Longitude	%Silt	%VFS	%Sand	%Clay	%Modified silt	%OC	Adj. %OC	Texture	Perme-ability Class	Structure Code	**Erodibility (K) value**
27	-28.87274	30.02662	18.03	7.66	10.15	64.15	25.69	3.75	3.75	Cl	6	2	0.153
28	-28.87383	30.02640	17.34	7.33	11.18	64.15	24.67	1.24	1.24	Cl	6	4	0.252
29	-28.87129	30.02249	25.91	13.58	41.98	18.52	39.49	1.60	1.60	SaLo	2	2	0.254
30	-28.87227	30.02998	15.62	11.46	17.48	55.44	27.08	3.19	3.19	Cl	6	3	0.221
31	-28.86909	30.01049	16.40	6.84	17.74	59.03	23.23	5.63	4.00	Cl	6	1	0.111
32	-28.86799	30.00883	21.51	9.59	16.79	52.11	31.10	2.43	2.43	Cl	6	4	0.294
33	-28.86842	30.00686	11.29	10.43	46.37	31.90	21.73	1.16	1.16	SaClLo	4	2	0.156
34	-28.87057	29.99986	21.06	9.21	23.88	45.85	30.27	2.00	2.00	Cl	6	4	0.312
35	-28.87028	29.99329	12.53	9.69	22.63	55.15	22.22	4.59	4.00	Cl	6	3	0.200
36	-28.87723	29.99339	14.35	10.69	32.62	42.33	25.05	3.31	3.31	Cl	6	3	0.238
37	-28.87723	29.99339	18.99	8.15	18.25	54.60	27.14	5.47	4.00	Cl	6	3	0.215
38	-28.86281	30.00006	24.11	11.56	36.86	27.46	35.67	2.83	2.83	SaClLo	4	2	0.230
39	-28.86153	30.00044	20.85	14.78	33.95	30.42	35.63	1.48	1.48	SaClLo	4	2	0.248
40	-28.85639	29.98362	12.93	24.83	41.49	20.76	37.75	1.20	1.20	SaClLo	4	2	0.307
41	-28.86019	29.99014	17.78	6.50	69.59	6.13	24.28	0.64	0.64	LoSa	2	1	0.136
42	-28.86008	29.99181	12.88	9.96	60.68	16.48	22.84	1.04	1.04	SaLo	2	1	0.091
43	-28.85896	30.01093	15.17	10.29	30.92	43.61	25.47	2.95	2.95	Cl	6	3	0.241
44	-28.84635	30.00047	17.93	20.27	46.43	15.37	38.20	1.24	1.24	SaLo	2	2	0.265
45	-28.84656	29.99500	17.78	9.98	33.05	39.19	27.76	2.00	2.00	ClLo	4	2	0.165
46	-28.84436	30.00060	21.74	16.23	39.82	22.20	37.97	1.88	1.88	SaClLo	4	2	0.286
47	-28.84126	30.00114	21.49	14.38	33.65	30.47	35.87	2.39	2.39	SaClLo	4	2	0.231
48	-28.84293	30.00393	17.19	13.21	46.48	23.12	30.40	1.36	1.36	SaClLo	4	2	0.237
49	-28.83948	30.00358	15.31	12.91	61.11	10.67	28.22	1.40	1.40	SaLo	2	1	0.146
50	-28.83828	30.00530	17.16	4.58	68.76	9.50	21.74	1.48	1.48	SaLo	2	1	0.090
51	-28.83765	30.01445	12.87	13.62	62.28	11.22	26.50	1.24	1.24	SaLo	2	1	0.132
52	-28.85744	30.02468	14.50	7.53	11.95	66.02	22.03	3.79	3.79	Cl	6	2	0.142
53	-28.83459	29.99031	16.95	11.75	50.24	21.07	28.70	2.99	2.99	SaClLo	4	2	0.199

5.1.2.2 Selection of interpolation method

Because the erodibility values were representatives of sampled points, filling non-measured areas was undertaken using spatial interpolation technique.

From the three interpolation techniques of ArcView GIS (ESRI, 1996), namely, Spline, Inverse Distance Weighting (IDW), and Kriging, the former that has smoothing effect was eliminated from further comparison because it included negative values in its results. The IDW works based on the inverse of distance to a power parameter whereby the influence of a point decreases as the distance form a cell increases (ESRI, 1996). In this technique, combinations of three power parameters, namely, ½, 2, and 3, with three search points, namely, 12, 24, and 38 using nearest neighbors were tested.

Kriging, is a widely employed method because of its accountancy for spatial correlation between points, preliminary production of semivariogram to indicate the extent of the dependence of the variance of values on the distance between points, and provision of estimation errors (Burrough and McDonell, 1998). As in IDW, the sample counts considered are 12, 24 and 38. The lag distance that represents the distance between two neighboring points is taken as 125 m, which is the minimum between any two points (Ahmed, 2003).

Comparing the interpolation techniques (Table 5.5), all the IDW methods with power parameter of ½ resulted in values that had minimum and maximum values, which could not include the minimum, and maximum values of the sample. For example, the interpolated K values from search point 12 with power ½ have minimum K value of 0.160 from which K values of 12 soil samples fall below this minimum value. The maximum value of this interpolation, 0.323, is less than K values from 2 soil samples. In ordinary kriging, similar results encountered with Circular, Gaussian, and Linear with Sill techniques for all search points. Further comparison of the interpolation techniques using standard deviation of residuals between the observed and the estimated values, R, and the goodness of prediction, G (Equation 5.2), produced results listed in the same table. R value close to zero indicates that the estimated value is well compared with the observed, whereas G value, which compares the goodness of spatially interpolated value with the one that would be estimated from the mean value, has high value for the best estimation. Based on these comparisons, the results (Table 5.5) show that exponential method of Ordinary Kriging with 24 sample counts is the best interpolation technique and hence its selection.

$$G = \left(1 - \left\{\sum_{i=1}^{n}\left[z(x_i) - \hat{z}(x_i)\right]^2 / \sum_{i=1}^{n}\left[z(x_i) - \bar{z}\right]^2\right\}\right)100 \qquad (5.2)$$

where $\bar{z}$ is the sample mean, $z(x_i)$ is the observed value at location i, $\hat{z}(x_i)$ is the predicted value at location i, and n is the sample size (Agterberg, 1989, cited in Schloeder *et al* (2001)).

Table 5.5 *Comparison of interpolation techniques for K surface derivation*

Interpolation type			Min.	No. Samples[a] < Min.	Max.	No. Samples[a] > Max.	R[b]	G[c]
	Search Point	**Power**						
IDW	12	½	0.160	12	0.323	2	0.075	11.02
		2	0.111	none	0.422	none	0.072	29.94
		3	0.111	none	0.422	none	0.078	13.36
	24	½	0.177	15	0.294	7	0.077	7.70
		2	0.111	none	0.422	none	0.071	25.74
		3	0.111	none	0.422	none	0.078	13.35
	38	½	0.187	16	0.277	7	0.078	4.33
		2	0.111	none	0.422	none	0.071	26.34
		3	0.111	none	0.422	none	0.078	14.24
	Method	**Sample Count**						
Ordinary Kriging	Circular	12	0.181	15	0.263	8	0.080	-3.28
		24	0.183	15	0.263	8	0.080	-3.53
		38	0.181	15	0.264	8	0.081	-4.52
	Exponential	12	0.115	none	0.417	none	0.061	11.02
		24	***0.115***	***none***	***0.417***	***none***	***0.063***	***36.28***
		38	0.115	none	0.417	none	0.064	34.63
	Gaussian	12	0.137	4	0.361	2	0.065	32.94
		24	0.140	4	0.357	2	0.068	28.77
		38	0.141	4	0.358	2	0.069	27.16
	Linear with Sill	12	0.182	15	0.260	8	0.080	-3.56
		24	0.182	15	0.260	8	0.080	-3.43
		38	0.175	14	0.261	8	0.081	-4.28
	Spherical	12	0.119	none	0.404	none	0.065	32.56
		24	0.119	none	0.404	none	0.065	33.59
		38	0.119	none	0.404	none	0.068	29.81

[a]Minimum and maximum values of K for the samples that were used to execute the interpolation are 0.111 and 0.422, respectively.
[b]**R** is standard deviation of residuals
[c]**G** is the goodness of prediction:

5.1.2.3 Creating K factor surface

All 53 point K values were then interpolated using the exponential kriging interpolation to create a continuous surface (Figure 5.5). The minimum and maximum values of the K surface are 0.105 and 0.395, respectively, which are acceptably comparable with the corresponding values of the samples. The mean value is 0.218 where as the coefficient of variation is 13.8%. The K variance ranges form 0.001-0.005 with mean value of 0.004 and standard deviation of 0.001. Looking at the spatial distribution, the highest K values are indicated in the central part; the medium values dominate the reserve except the northern part represented by low values. The spatial extent of each class as reported in the figure is given in Table 5.6.

Table 5.6 *Erodibility values and their areal extent of WGR*

Erodibility class	% Area
0.108-0.137	0.3
0.137-0.169	4.7
0.169-0.201	24.2
0.201-0.233	38.7
0.233-0.266	27.6
0.266-0.298	3.2
0.298-0.330	0.8
0.330-0.362	0.3
0.362-0.395	0.1

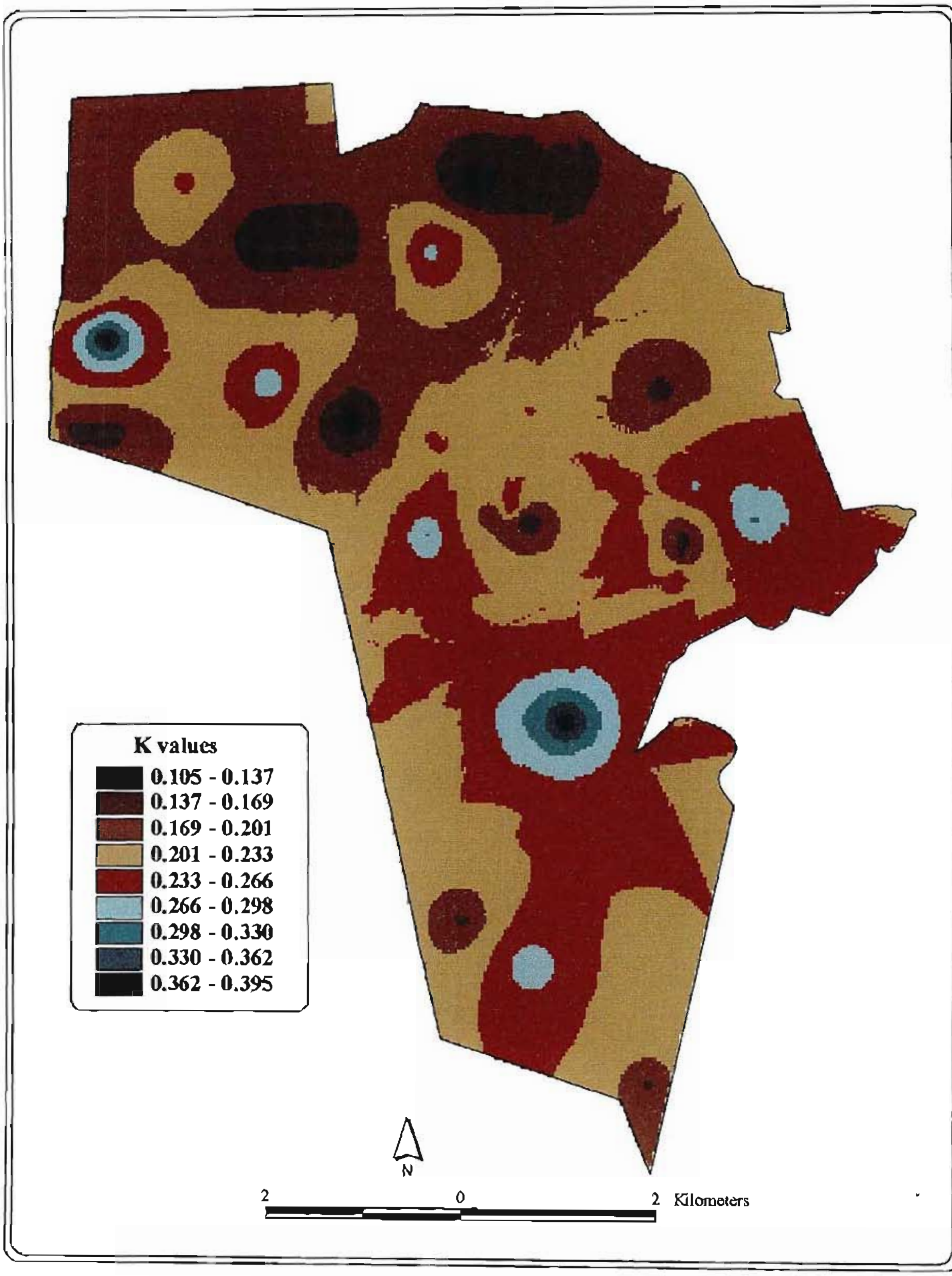

Figure 5.5 *K factor map of WGR*

5.1.3 Topographic factor (LS)

5.1.3.1 Elevation

After extracting the DEM within the study area, the minimum and maximum altitude of the area are 890 and 1310 m amsl, respectively (Figure 5.6), which is comparable to the observed minimum and maximum altitudes. The mean elevation is 1061 m amsl with standard deviation of 98. Dividing the elevation classes into three sections (Table 5.7), the relatively lowest areas (890-1024 m amsl) are located in the northern part extending along the eastern strip, and along the Bushman's River and the south eastern strip; medium altitude areas (1024-1164 m amsl) are mostly found in the central part of the reserve on either side of the Bushman's River, the majority being found north of the River; and high altitude areas (1164-1310 m amsl) are observed north of the Bushman's River topping the medium altitude areas and in the southern tip of the reserve.

Table 5.7 *Altitude classes and their areal extent of WGR*

DEM class (m amsl)	% Area
890-1026	46.5
1026-1168	34.6
1168-1310	18.9

5.1.3.2 Slope

The slope map (Figure 5.7), which is an indication of steepness, was derived from the DEM and ranges from 0% (0°) to about 133% (53°) (Table 5.8). The mean slope of the reserve is about 14% (8°) and the standard deviation is 16% (9°). Areas bordering the Bushman's River and extending northeast and southeast of either side are the steepest (18-90%), whereas the northern areas have relatively flat to gentle slope (0-5%). Nearly 81% of the grids have slopes less than 25% (14°).

Table 5.8 *Slope classes and their areal extent of WGR*

Slope class (° / %)	% Area
0-6 / 0-10.5	53.1
6-12 / 10.5-21.3	22.6
12-18 / 21.3-32.5	10.2
18-24 / 32.5-44.5	4.8
24-30 / 44.5-57.7	4.5
30-36 / 57.7-72.7	2.4
36-42 / 72.7-90.0	1.6
42-48 / 90.0-111.1	0.5
48-53 / 111.1-132.7	0.1

5.1.3.3 Flow accumulation

The first step in determining flow accumulation is identifying the direction of flow (Figure 5.8) from one pixel to the nearest neighbor(s) (ESRI, 1996), which works based on the principle that water flows from one grid cell to the next following the steepest descent as calculated by Equation 5.3 (ESRI, 1996). This steepest descent is the result of elevation comparison of a 3 by 3 matrix of cells whereby 8 cells surround the cell that releases flow.

$$\text{Slope} = (\text{drop in } z \,/\, \text{distance}) \times (100) \tag{5.3}$$

where z is elevation of the grid cell and distance is taken between the centers of neighboring cells.

Flow accumulation, then, uses flow direction as input for its derivation and is given as the number of all cells that drain into an output cell (ESRI, 1996). Since this output (Figure 5.9) is a network of cells over which water flows, the number of cells that flow into this "channel" is high. This value should, therefore, be modified to conform to the RUSLE threshold values (Engel, 2003; Mitasova, 2004), which is taken in this study a maximum of 90 m due to the undulating nature of the landscape. Since the side of a grid cell is specified at 30 m, the maximum slope length represents 3 cells whereas the minimum is 1 cell where flow will be initiated. The modified flow accumulation (Figure 5.10) has mean value and standard deviation of 2.49 and 0.86, respectively. Majority of flow accumulation (73.9%) has values between 2 to 3 cells (60 to

90 m) (Table 5.9). The flow accumulation map indicates that all values are distributed throughout the reserve.

Table 5.9 *Flow accumulation values and their areal extent of WGR*

Flow accumulation class (cells)	% Area
1-2	26.1
2-3	73.9

5.1.3.4 Topographic factor (LS)

Combining the slope and the flow accumulation within ArcView GIS using Equation 4.6 gave the LS factor surface map shown as Figure 5.11 with values ranging from 0 to 62. The mean value is 5 whereas standard deviation is 7, thus, the variation of the grid cells in their value of this factor is high. From the reclassified class intervals (Table 5.10), the majority of the area (72.1%) falls below the mean value. Most of the high values are noticed along the steepness of the Bushman's River and the escarpments that extend northeast and southeast. In general, the LS value might appear large which is expected of low resolution DEMs that tend to miss physical barriers within the specified grids, which would reduce runoff (Hickey *et al*, 1994). Since the LS factor is the product of slope length (flow accumulation) and slope steepness, its proportional values to these inputs are directly related.

Table 5.10 *Combined LS values and their areal extent of WGR*

LS class	% Area
0-3	51.2
3-7	20.9
7-12	9.4
12-18	6.4
18-25	4.7
25-35	4.1
35-45	2.3
45-62	0.9

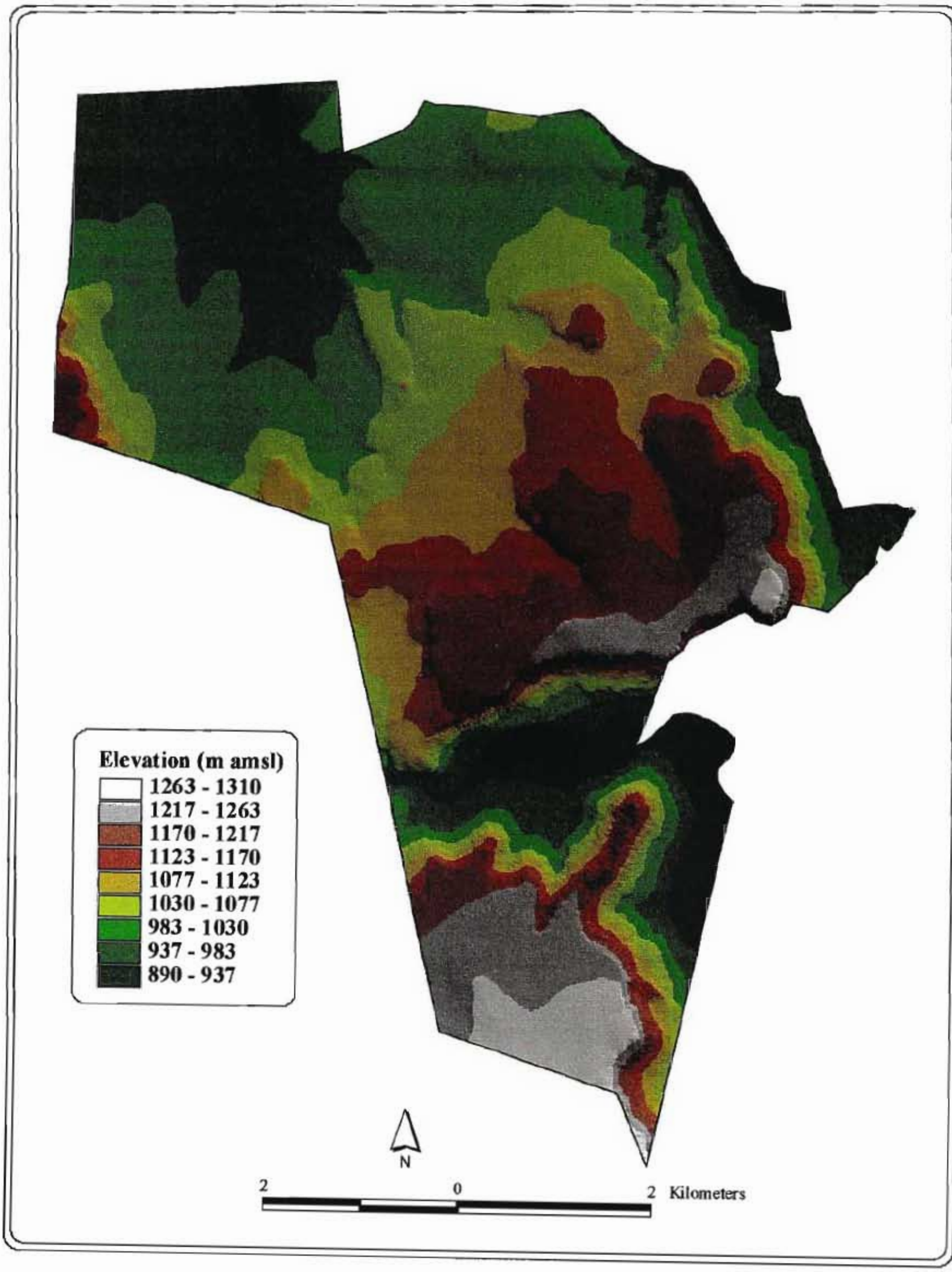

Figure 5.6 *Triangulated Irregular Network showing elevations of WGR*

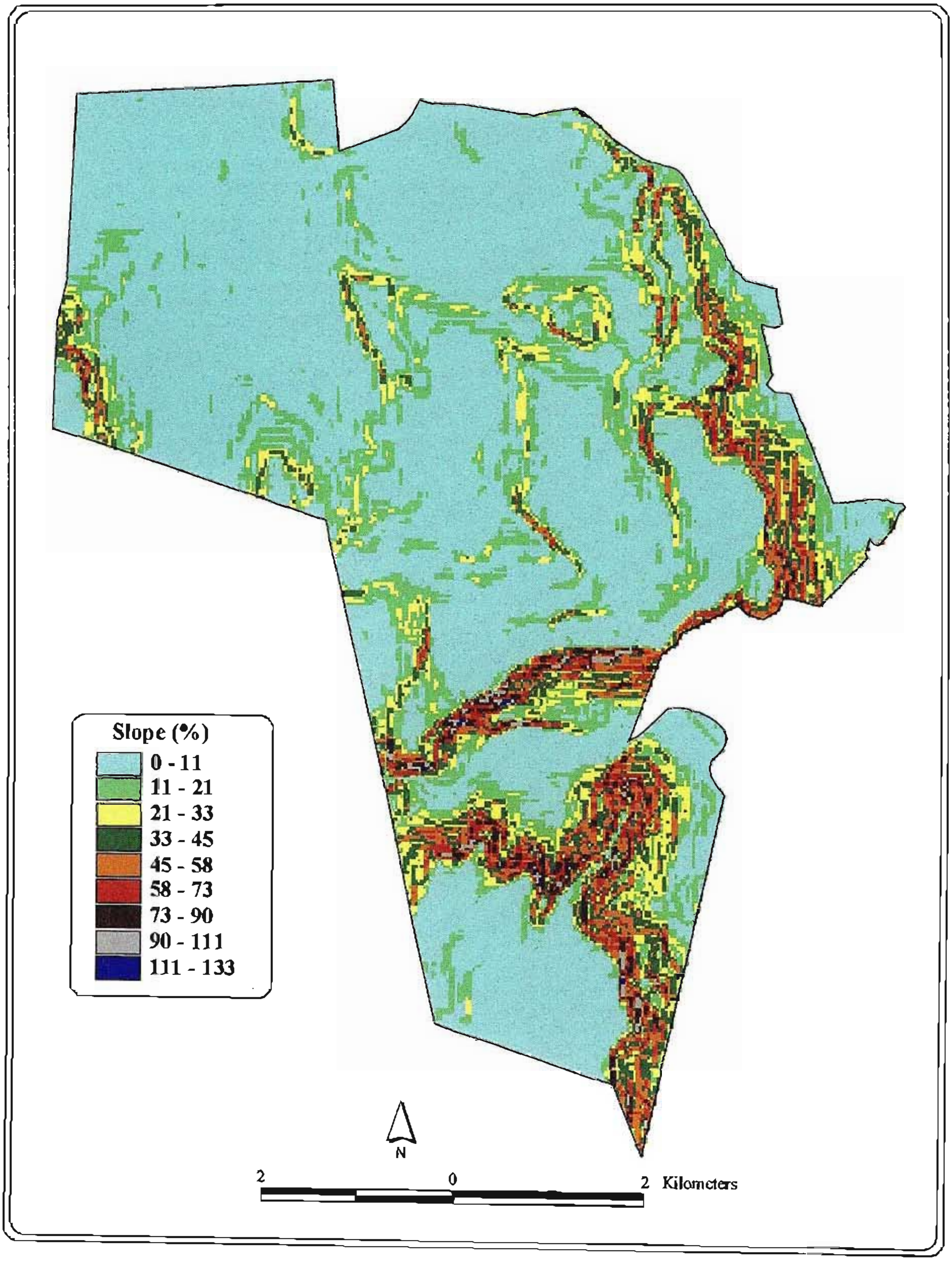

Figure 5.7 *Slope map of WGR*

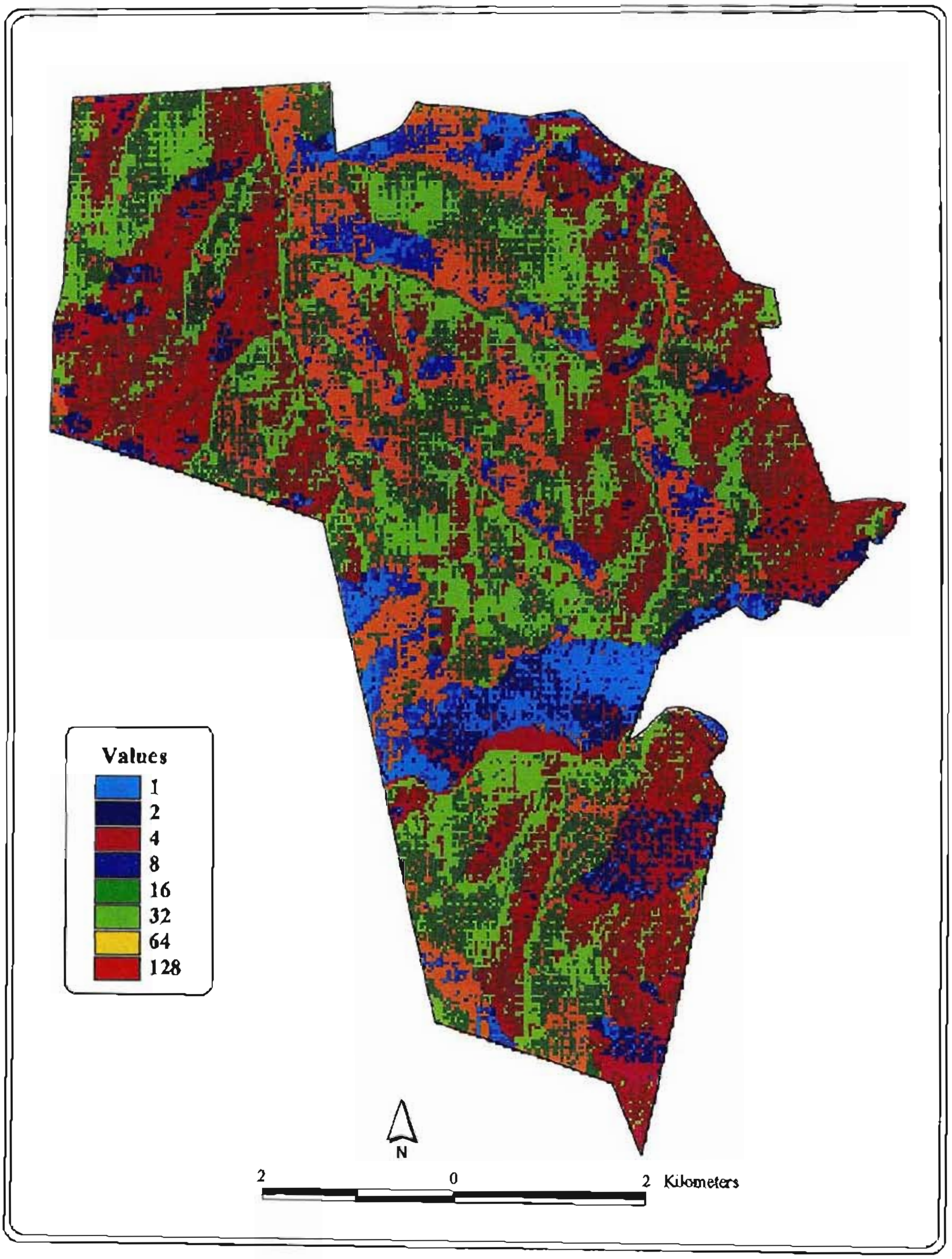

Figure 5.8 *Flow direction map of WGR*

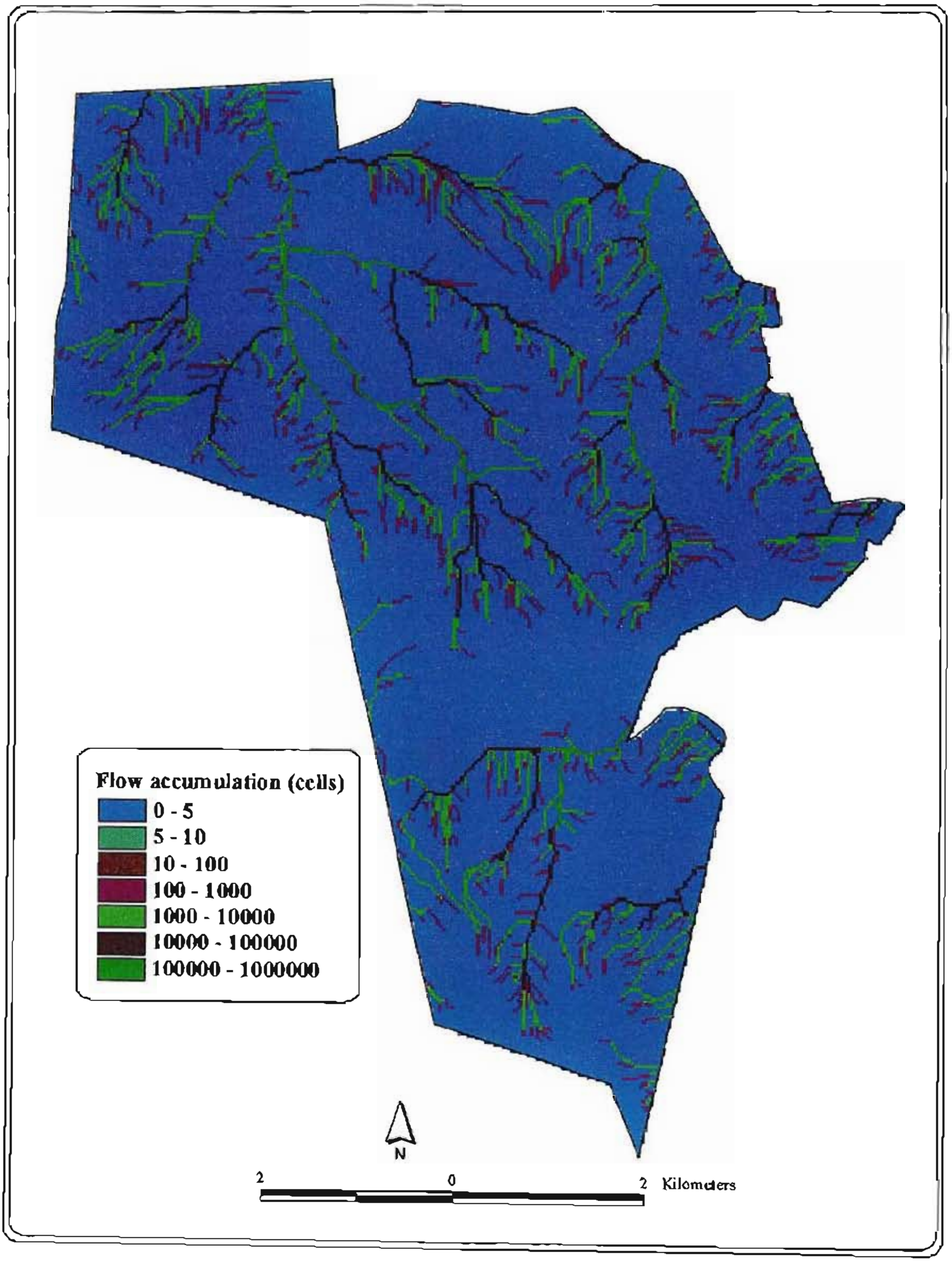

Figure 5.9 *Original flow accumulation map of WGR*

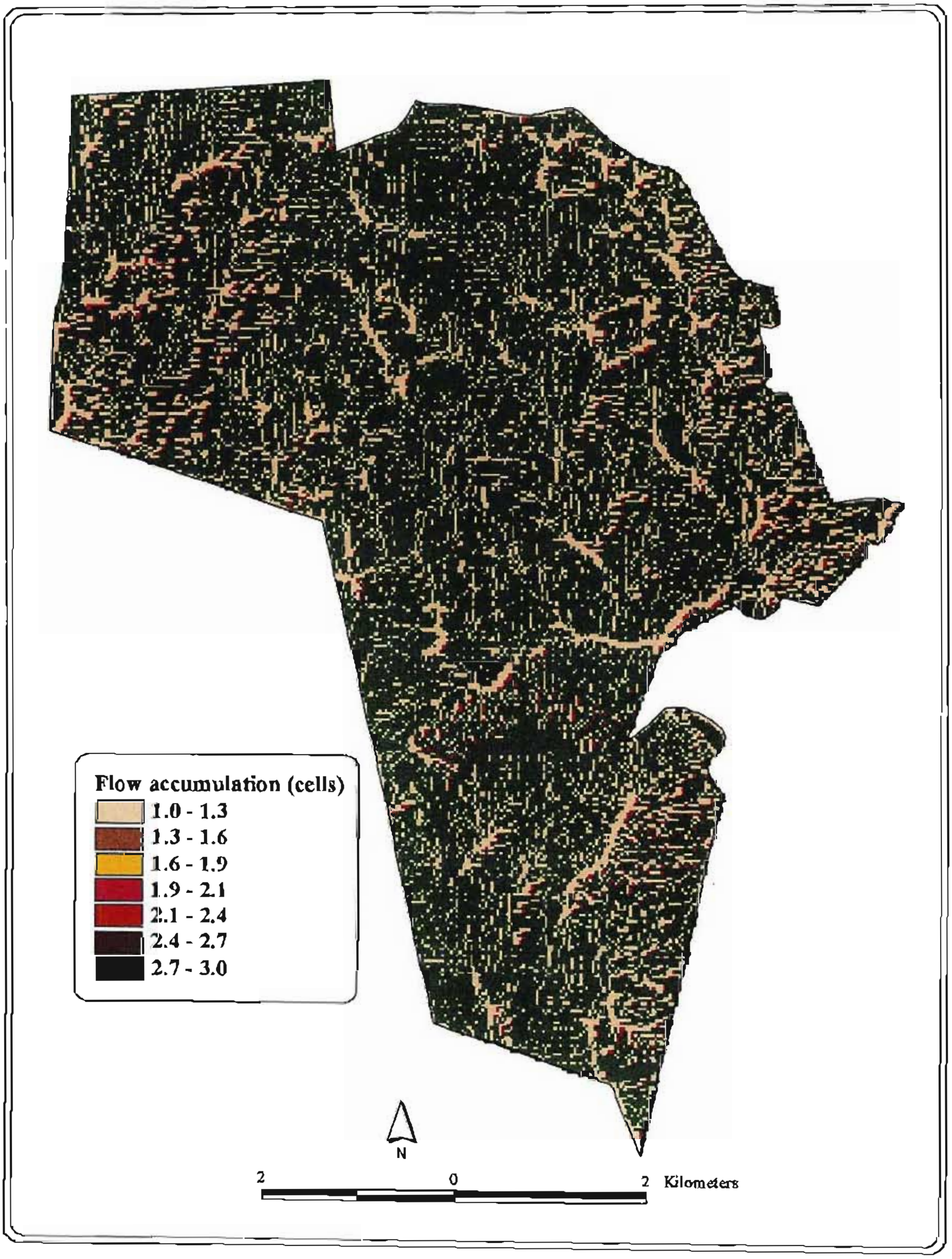

Figure 5.10 *Modified flow accumulation map of WGR*

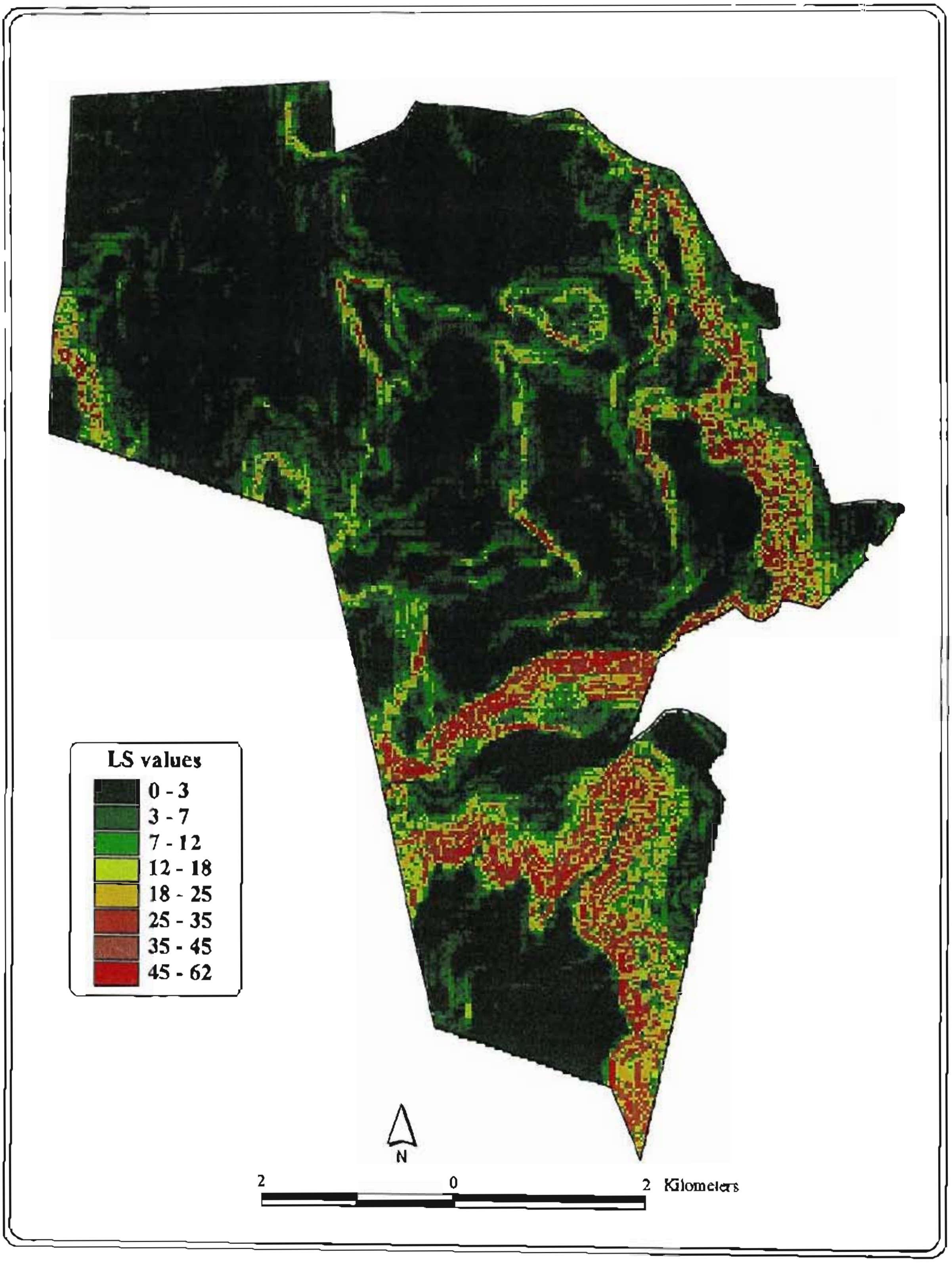

Figure 5.11 *LS factor map of WGR*

5.1.4 Cover management factor (C)

The principal parameter used in this research to assign the C value was % ground cover. The number of cover counts using the point-intercept method and their % values are given in Table 5.11. Subsequent assigning of C values from the USLE table (Table 5.12) (Wischmeier and Smith, 1978) to the % covers gave results listed in column 4 whereas the average pixel readings from the 5 vegetation indices are given in the latter columns of the same table. The USLE table of C values provides a limited number of % covers and hence in this research, occasionally, it is tried to assign values by inferring from the given ones. The minimum value given is 0.003 when the land has nearly complete (more than 95%) vegetation cover. In contrast, the maximum value is 0.15 given to the poorest land cover in the area.

Remote sensing application in cover factor derivation is based on the vegetation cover of the area. Since the effect of land cover in soil erosion assessment is on interception of raindrop and obstruction of surface flow, it is the spectral response of vegetation cover that is required to quantify cover amount. However, in vegetation-sparse areas, the interference of soil or bare ground is considerable that reduces the capture of reflectance of vegetation (Tueller, 1987). This problem is reduced by applying vegetation indices that maximizes the visibility of vegetation and minimizes the effect of soil background (Ehlschlaeger and Tweddale, 2000).

A fixed factor (for example, 0.5 for intermediate vegetation cover) is used in Equation 4.9 for SAVI. The slope (a) and intercept (b) parameters of the TSAVI (Equation 4.10) are extracted from the soil line (Figure 5.12) that was plotted on the x and y axes representing Red and NIR band reflectance values, respectively. From the line, represented by Equation 5.4, a is found to be 0.8602 and b is 9.2153.

$$NIR = (0.8602)(Red) + 9.2153 \tag{5.4}$$

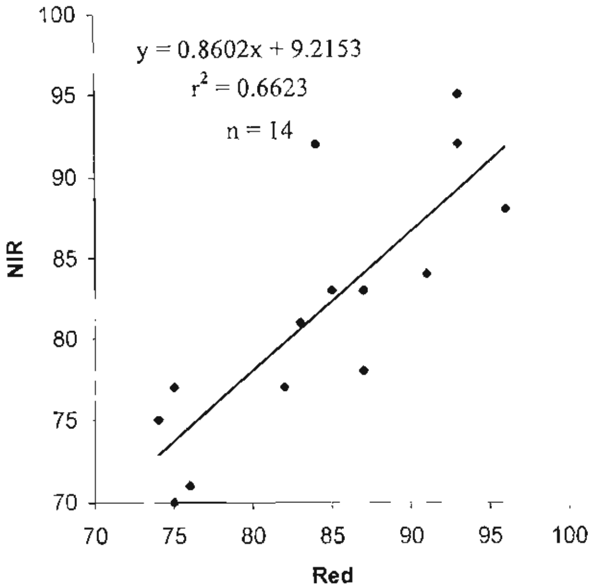

Figure 5.12 *Bare soil line curve*

For all indices quadratic equations using multiple linear regression gave the best fitness between the pixel values of the vegetation indices and the calculated C values. These are:

$$USLE\_C = (0.746)(NDVI)^2 - (0.768)(NDVI) + 0.210 \quad (5.5)$$

$$USLE\_C = (2.120)(TNDVI)^2 - (4.350)(TNDVI) + 2.240 \quad (5.6)$$

$$USLE\_C = (0.339)(SAVI)^2 - (0.520)(SAVI) + 0.212 \quad (5.7)$$

$$USLE\_C = (0.875)(TSAVI)^2 - (0.837)(TSAVI) + 0.214 \quad (5.8)$$

$$USLE\_C = (0.357)(MSAVI)^2 - (0.578)(MSAVI) + 0.237 \quad (5.9)$$

The coefficients of determination, r^2, and the standard errors of prediction, s.e, based on the mean values of both the independent (the indices as well as their squares) and dependent (estimated C) variables are given in Table 5.13. Comparisons of the estimated against the calculated values after running the vegetation index pixel values in the respective equations (Equations 5.5-5.9) are illustrated as Figure 5.13.

Table 5.11 *Quantitative mulch estimates, USLE C readings, and pixel values*

Sample	No. of counts	% Cover	USLE C	NDVI	TNDVI	SAVI	TSAVI	MSAVI
1	16	53.3	0.038	0.338	0.915	0.505	0.325	0.501
2	14	46.7	0.042	0.382	0.938	0.571	0.367	0.549
3	17	56.7	0.040	0.345	0.919	0.515	0.329	0.511
4	19	63.3	0.080	0.229	0.854	0.342	0.216	0.370
5	12	40.0	0.150	0.173	0.819	0.258	0.165	0.285
6	13	43.3	0.090	0.219	0.848	0.327	0.207	0.357
7	24	80.0	0.070	0.297	0.892	0.444	0.280	0.454
8	24	80.0	0.070	0.297	0.892	0.444	0.280	0.454
9	Estimated	95.0	0.003	0.477	0.988	0.712	0.456	0.644
10	26	86.7	0.013	0.465	0.982	0.694	0.445	0.632
11	14	46.7	0.090	0.184	0.827	0.274	0.170	0.308
12	23	76.7	0.040	0.246	0.863	0.368	0.229	0.391
13	21	70.0	0.030	0.290	0.889	0.433	0.270	0.447
14	17	56.7	0.038	0.352	0.923	0.527	0.333	0.519
15	29	96.7	0.003	0.508	1.004	0.758	0.484	0.672
16	13	43.3	0.050	0.266	0.875	0.398	0.257	0.418
17	Estimated	40.0	0.013	0.430	0.965	0.643	0.413	0.600
18	17	56.7	0.042	0.335	0.914	0.500	0.319	0.500
19	19	63.3	0.042	0.344	0.918	0.514	0.313	0.507
20	14	46.7	0.080	0.328	0.910	0.490	0.314	0.492
21	13	43.3	0.100	0.171	0.819	0.255	0.156	0.290
22	18	60.0	0.038	0.356	0.925	0.531	0.335	0.522
23	29	96.7	0.042	0.331	0.911	0.494	0.313	0.495
24	24	80.0	0.035	0.354	0.924	0.529	0.335	0.521
25	17	56.7	0.030	0.393	0.945	0.587	0.378	0.562
26	20	66.7	0.038	0.379	0.935	0.566	0.352	0.536
27	10	33.3	0.100	0.188	0.829	0.282	0.183	0.310
28	13	43.3	0.080	0.164	0.814	0.245	0.154	0.278
29	28	93.3	0.011	0.352	0.923	0.526	0.328	0.519
30	17	56.7	0.050	0.312	0.901	0.467	0.301	0.474
31	14	46.7	0.050	0.312	0.901	0.466	0.299	0.472
32	13	43.3	0.050	0.278	0.882	0.415	0.257	0.432
33	12	40.0	0.050	0.276	0.881	0.412	0.261	0.430
34	29	96.7	0.011	0.450	0.975	0.672	0.429	0.619
35	23	76.7	0.012	0.253	0.868	0.379	0.241	0.403
36	20	66.7	0.032	0.369	0.932	0.551	0.355	0.536
37	Estimated	90.0	0.003	0.308	0.899	0.461	0.294	0.470
38	22	73.3	0.036	0.364	0.929	0.544	0.349	0.532
39	9	30.0	0.150	0.087	0.765	0.130	0.091	0.153
40	15	50.0	0.060	0.276	0.880	0.412	0.258	0.428
41	16	53.3	0.042	0.390	0.943	0.583	0.375	0.558
42	27	90.0	0.012	0.436	0.968	0.652	0.411	0.606
43	18	60.0	0.032	0.477	0.988	0.713	0.457	0.644
44	24	80.0	0.012	0.341	0.917	0.509	0.323	0.507
45	22	73.3	0.012	0.436	0.967	0.652	0.417	0.605

Table 5.12 *C-factor for permanent pasture, veld, and woodland* (Wischmeier and Smith, 1978)

Vegetation canopy	Canopy cover (%)	Cover that contacts the soil surface						
		Type	% ground cover					
			0	20	40	60	80	95+
No appreciable canopy	25	G	0.45	0.20	0.10	0.042	0.013	0.003
		W	0.45	0.24	0.15	0.091	0.043	0.011
Grassland or short brush with average drop height of 0.5m	25	G	0.36	0.17	0.09	0.038	0.013	0.003
		W	0.36	0.17	0.13	0.083	0.041	0.011
	50	G	0.26	0.13	0.07	0.035	0.012	0.003
		W	0.26	0.16	0.11	0.076	0.039	0.011
	75	G	0.17	0.10	0.06	0.032	0.011	0.003
		W	0.17	0.12	0.09	0.068	0.038	0.011
Appreciable brush or bushes, with average drop fall height of 2m	25	G	0.40	0.18	0.09	0.040	0.013	0.003
		W	0.40	0.22	0.14	0.087	0.042	0.011
	50	G	0.34	0.16	0.08	0.038	0.012	0.003
		W	0.34	0.19	0.13	0.082	0.041	0.011
	75	G	0.28	0.14	0.08	0.036	0.012	0.003
		W	0.28	0.17	0.12	0.078	0.040	0.011
Trees, but no appreciable low brush. Average drop fall height of 4m	25	G	0.42	0.19	0.10	0.041	0.013	0.003
		W	0.42	0.23	0.14	0.089	0.042	0.011
	50	G	0.39	0.18	0.09	0.040	0.013	0.003
		W	0.39	0.21	0.14	0.087	0.042	0.011
	75	G	0.36	0.17	0.09	0.039	0.012	0.003
		W	0.36	0.20	0.13	0.084	0.041	0.011

Table 5.13 *Comparisons of vegetation indices using selected statistical parameters*

Vegetation index	Coefficient of determination (r^2)	Standard error of prediction (s.e.)	Correlation coefficient (r)
NDVI	0.71	0.0192	0.85
TNDVI	0.71	0.0190	0.85
SAVI	0.71	0.0192	0.85
TSAVI	0.70	0.0194	0.85
MSAVI	0.72	0.0188	0.86

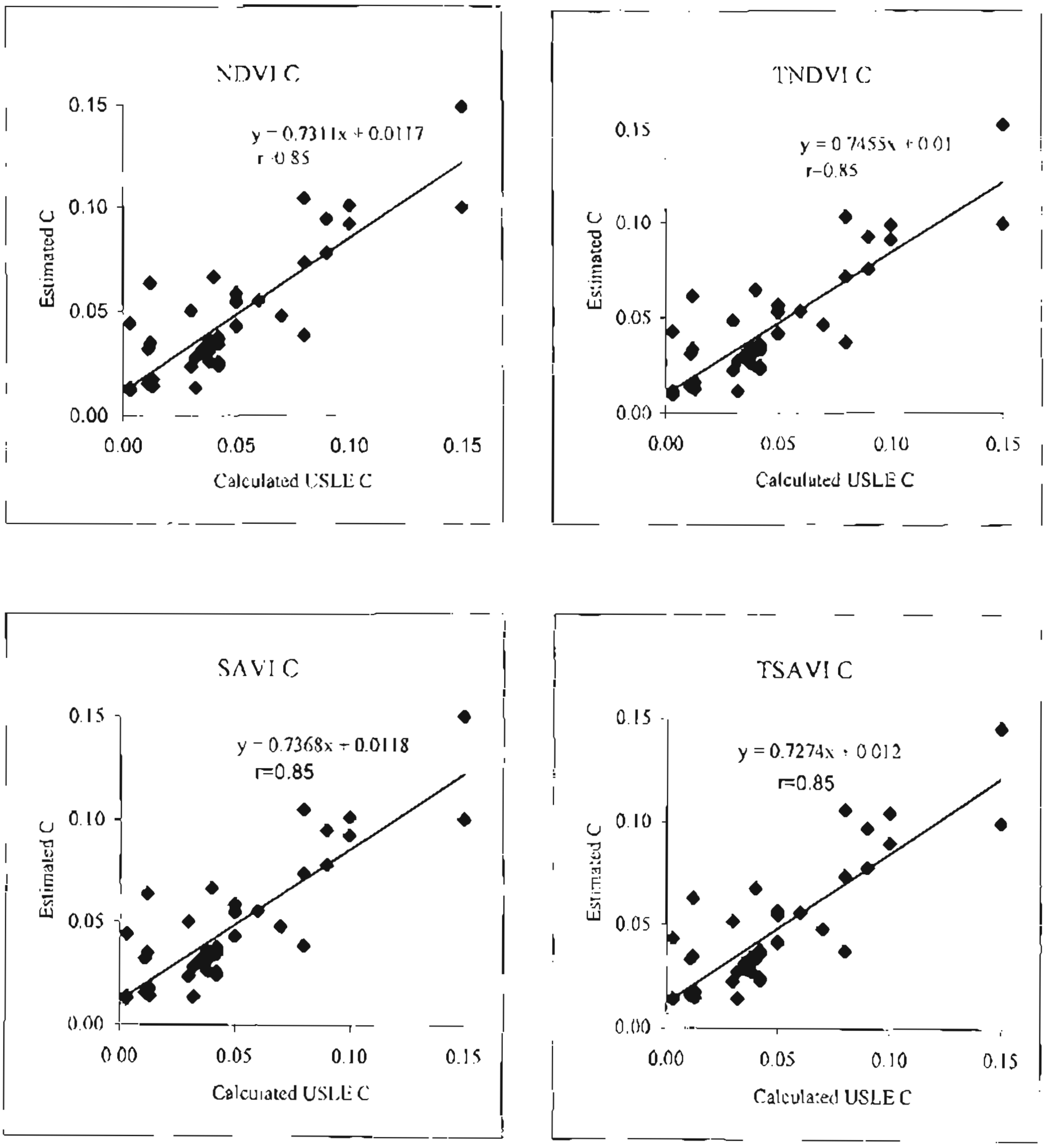

Figure 5.13 *Fitness graphs of vegetation derived C values with the calculated USLE C values*

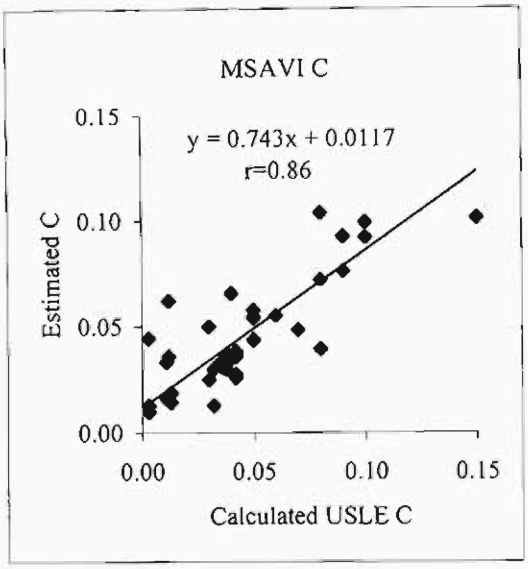

Figure 5.13 *Continued*

Evaluation of each derived image against the calculated C values based on basic statistical comparisons is summarized in Table 5.14. The lowest C value, 0.004, is observed in the MSAVI, which is comparable to 0.003 of the observed followed by TNDVI, 0.009. Note from the Table that the maximum values (column 4) of the population pixels in all indices are large compared to the values of sample pixels. However, their effect on other statistical descriptions is minimal implying that they have insignificant spatial frequency (column 6). If comparison of results from statistical descriptions of Table 5.13 and Table 5.14 at any scale is needed, the MSAVI is the best.

The final C factor map (Figure 5.14) was then produced using the MSAVI. After reclassifying the intervals as given in the figure, the spatial extent of each class interval is summarized in Table 5.15. About 72% of the area has C factors that range from 0.01 to 0.06 covering almost the whole southern section of the reserve except the southern plateau, and most of the northern section. The remaining area, predominantly the northwestern part and the southern plateau, have C factor values of 0.06 or more.

Table 5.14 *Statistical comparisons of vegetation indices derived C values*

Vegetation index	Mean	Minimum	Maximum	Standard deviation	% Frequency
Calculated C	0.046	0.003	0.150	0.035	
NDVI	0.046*	0.012*	0.149*	0.030*	
	0.047	0.012	0.403	0.034	1.9
TNDVI	0.045*	0.010*	0.153*	0.030*	
	0.046	0.009	0.511	0.035	2.1
SAVI	0.046*	0.012*	0.150*	0.030*	
	0.048	0.013	0.407	0.034	2.0
TSAVI	0.046*	0.014*	0.145*	0.030*	
	0.048	0.014	0.458	0.034	2.2
MSAVI	0.047*	0.010*	0.157*	0.030*	
	0.048	0.004	0.632	0.036	1.9

*values for samples; the rest are for population.

Table 5.15 *C values and their areal extent of WGR*

C class	% Area
0.004-0.005	0.1
0.005-0.006	0.2
0.006-0.008	0.8
0.008-0.010	1.3
0.010-0.030	33.1
0.030-0.060	38.7
0.060-0.100	19.4
0.100-0.200	5.7
0.200-0.632	0.8

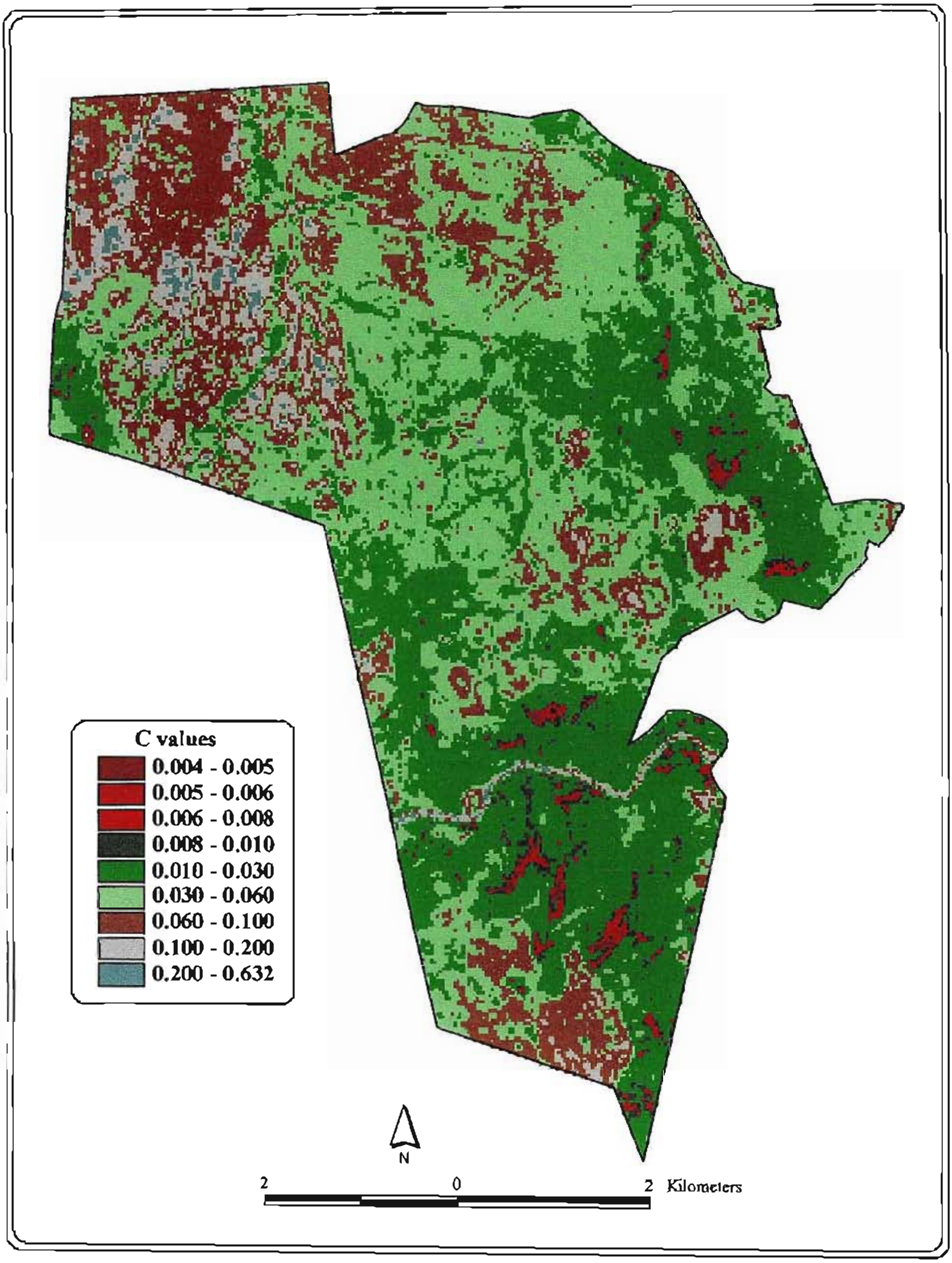

Figure 5.14 *C factor map of WGR*

5.1.5 Conservation practices factor (P)

The effect of conservation practice on soil loss is a reflection of such activities as terracing and contouring. Since WGR served in the past as a trial area for soil conservation structures, there are stone works, namely, stone terraces and check dams. These structures are limited to the northern section of the reserve. However, none of these structures are to be located in any of the sampled places and hence the P factor is assumed to be 1.

5.1.6 Soil loss map (A)

Annual soil loss map is the product of all the factor maps (Equation 2.2) and is given as in Figure 5.15. The minimum and maximum values of the potential soil losses are 0 and 346 ton ha^{-1} $year^{-1}$, respectively, and the potential mean soil loss is 5 ton ha^{-1} $year^{-1}$ whereas the standard deviation is about 12.5 ton ha^{-1} $year^{-1}$. More than 57% of the area has potential soil loss of less than or equal to 1 ton ha^{-1} $year^{-1}$ whereas about 75% has less than the average soil loss (Table 5.16). The low levels of soil losses are dominant in all parts of the reserve except along the sides and ridges of the Bushman's River that extend northeast and southeast, and scattered in the central and north western end of the northern section that have soil loss of more than the average.

Table 5.16 *Soil loss values and their areal extent of WGR*

Soil loss class (ton ha^{-1} $year^{-1}$)	Area (ha)	% Area
0-5	3642	74.6
5-20	923	19.1
20-50	236	4.9
50-346	54	1.1

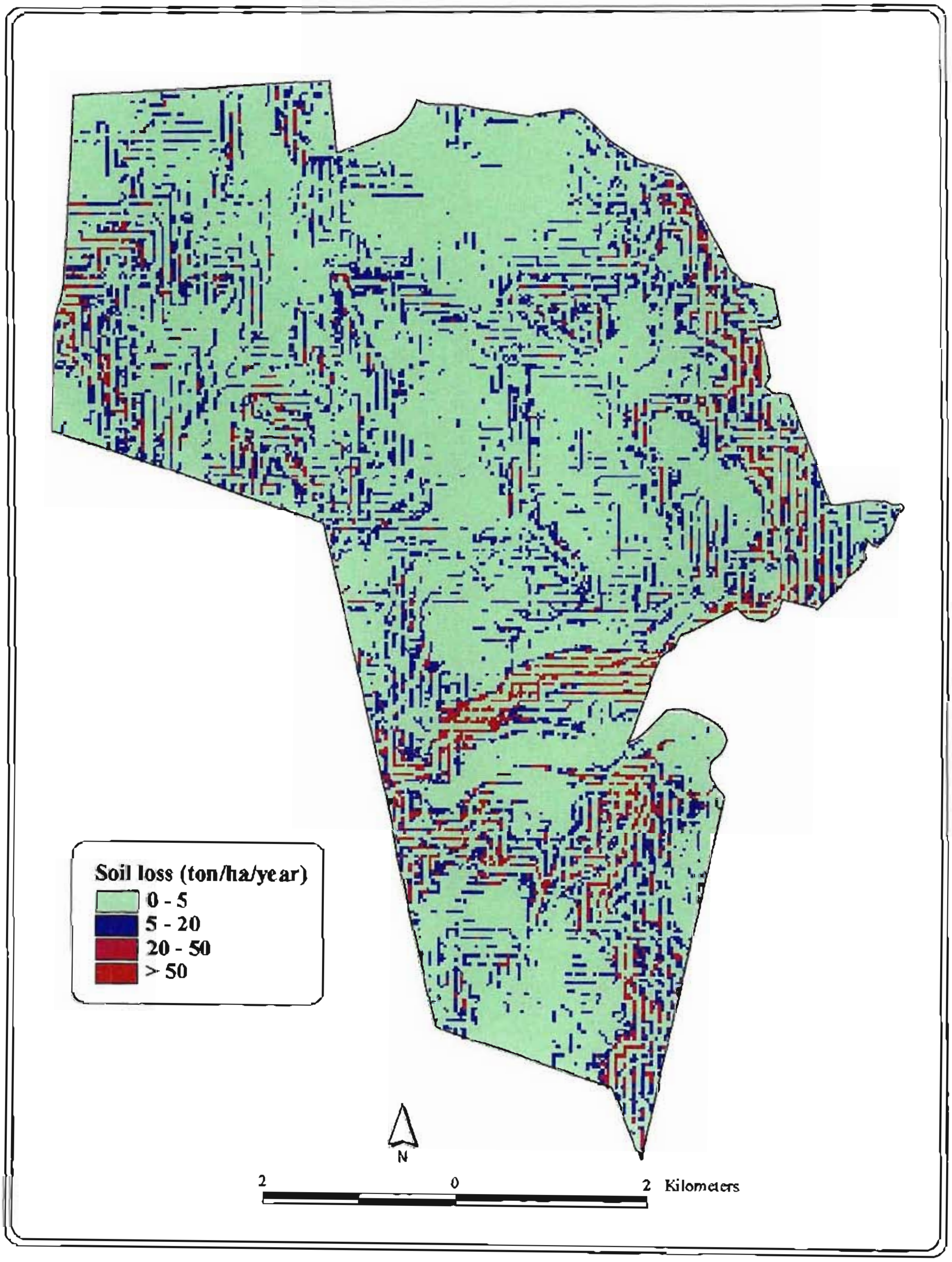

Figure 5.15 *Mean annual potential soil loss map of WGR*

5.2 DISCUSSION

The distribution of rainfall depth and EI_{30} depicts high seasonal variability as explained by the coefficients of variation. Within season variation is higher during the non-rainy season implying that rainfall in the area is more erratic during the dry season. However, the summer season share of EI_{30} is even higher than the summer rainfall depth. This is likely to be the result of the presence of a significant number of rainfall events with non-erosive magnitude in the drier season.

The K values, in general, are considered low. This assessment is based on the comparison of the results with soil erodibility values, which were derived form the soil erodibility nomograph, with qualitative meanings indicated in Table 5.17. From the comparison, the erodibility values fall under Very Low (25.9%), Low (64.8%), and Moderate classes (9.3%). The main reason for the low soil erodibility lies on the soil properties such as clay, and organic matter content, which are relatively high in the area. Such high organic matter and clay content were also reported by Ghebremeskel (2003).

Table 5.17 *ACRU erodibility rating* (Lorentz and Schulze, 1994)

Soil Erodibility Class	K nomograph
Very High	>0.70
High	0.50-0.70
Moderate	0.25-0.50
Low	0.13-0.25
Very Low	<0.13

Studying the slope of an area in soil erosion assessment is necessary since runoff, and soil detachment and movement increases directly with gradient. The slope of WGR has high variability indicating the difference in their effect on soil loss. Most of the central and the eastern parts are shown to have high values with their eminent contribution towards soil loss. In a separate evaluation of the flow accumulation, which has substituted flow length, its spatial distribution in the reserve is nearly similar, but the relatively flat areas have marginally higher values since such areas extend longer with uniform steepness. The final result that combined the effects of both slope and flow accumulation indicate that the highest effects are prevalent in steep areas. This suggests that slope rather than flow accumulation influences the topographic factor more.

The USLE C factor equations that were derived from the vegetation indices have shown strong statistical similarities among themselves. Such characteristics were also reported by Ehlschlaeger and Tweddale (2000), who found the MSAVI as the best fit with the USLE C factor. Wang *et al* (2002) used several band combinations including NDVI of Landsat TM image and the results showed strong similarity of the indices. Thus, the selection of MSAVI is only marginal implying that all the equations might result in nearly equally accepted maps. The resulting map (Figure 5.14) depicts that the natural vegetation condition is well reflected, which would otherwise be difficult using the ordinary statistical interpolation techniques that largely consider spatial proximity from the sample. The result has also enabled mapping inaccessible areas using the electromagnetic reflectance similarity of ground features through remote sensing.

High vegetation areas, corresponding to low C values, are found mostly along the valley through which the Bushman's River flows, the steep lands of the eastern escarpments, and along watercourses. The most likely reason for high vegetation cover along watercourses is the retention of moisture for comparatively longer time periods whereas the steep areas had less intervention by human activities in the past such as cultivation. The low vegetation covers, corresponding to high C values, are seen predominantly in the western part of WGR. Another important characteristic of the map is the wide variation of the C values with their non-uniformity in spatial distribution. The most likely cause could be the nature of rangelands that display a wide range of temporal and spatial vegetation growth discrepancy as pointed out by Abel and Stocking (1987) and Bartsch *et al* (2002).

Though the variation in soil loss in the area is very high, the majority of the area being dominated by low risk of soil losses that lie below the average might be considered promising. The average soil loss of the area, 5 ton ha^{-1} $year^{-1}$, compares with Martin (1987) who estimated 3.2 to 6.5 ton ha^{-1} year^{-1} for east coast catchment of KwaZulu-Natal. However, it is meaningful if soil loss estimation is compared with soil loss tolerance value (T), which is defined as "the maximum rate of soil erosion that can occur and still permit productivity to be sustained economically" (Renard *et al*, 1997, p.12), or, soil depths should be greater than rooting depth so that after soil loss the remaining soil should at least support rooting (Wight and Lovely, 1982). T value depends on the natural soil formation rates, which is estimated to be around 1 ton ha^{-1} year^{-1} (Stocking, 1884, cited in Martin (1987)), and on the soil depth; that is, higher soil formation rates are related to a higher T values, and deep soils have a higher permissible values than shallow ones.

There is a general agreement on the literature that T values on rangelands should be low. Wight and Lovely (1982) pointed that because of the general slow soil forming rates of arid and semi-arid rangelands, even smaller amount of soil loss could be serious as its impact is felt immediately by decrease in vegetation cover. These values are given in a range of 0 to 5 ton ha^{-1} year^{-1}. Scott (1951, cited in Scotney (1978)) suggest that well-managed natural veld experiences soil losses of less than 1 ton ha^{-1} year^{-1}, and as such losses are almost in equilibrium to natural soil loss rate. A comparable result from Botswana (Abel and Stoking, 1987) using the SLEMSA model showed that the annual soil loss on rangeland is less than 1 ton ha^{-1} year^{-1}. Realizing the critical soil depth condition of WGR, which mostly has soil depth of less than 300 mm (Hughes, 1989; Ghebremeskel, 2003), the T value should be set to a low value. Accepting a permissible geological soil loss of 0.16 ton ha^{-1} year^{-1} set in the Tugela basin (Murgatroyd, 1979) in which WGR is located, about 54% of the area experiences less than this rate of soil loss.

Looking at the spatial distribution of soil loss, though well-covered with vegetation, most of the steep areas of the central and the eastern escarpments have high erosion potential. The likely implication of this is the stronger relation of soil loss to topography rather than to vegetation cover. Moreover, the fact that the same areas are dominated by bare rocks might be an indication of extreme soil loss. This deduction conforms to the conclusion of Kienzle and Lorentz (1993) from their study in Henley Dam catchment, KwaZulu-Natal. Based on field plots study in Umfolozi Game Reserve (UGR), Venter *et al* (1989) found that topography rather than land use had significant effects on soil loss. Mentis (1982) also concluded that topography along with erodibility had the highest effect on soil loss calculated with the USLE. Similarly, from a rainfall simulator plot experiment in the University of the Free State, Snyman and van Rensburg (1986) reported a high runoff and soil loss from bare and steep treatments. Putting aside the effects of the factors of soil loss considered, absence or little disturbance of the surface can be ascribed as a major reason for this.

CHAPTER SIX: CONCLUSIONS AND RECOMMENDATIONS

WGR is an area that indicates the devastating effects of soil loss that took place in the past. This study was undertaken with the general objective of assessing potential soil loss in this area. Execution of the study followed the application of the most widely used RUSLE model. As reported in many instances, the capability of this model is enhanced due to its integration with GIS and remote sensing technologies. This approach was employed in this study and it was able to produce results that are likely to be of great importance to land managers in future environmental strategies. In this chapter the results are summarized briefly and conclusions based on the results are presented. Next, recommendations are given. The recommendations are made based on the severity of the problem, which differs spatially.

6.1 CONCLUSIONS

In general, the potential soil loss in the study area is high as compared to naturally occurring rates. However, the occurrence of soil losses of such (or at any) extent largely depends on the management of the reserve. Also, the highest erosion rates are noticed in steep areas, most of which have considerable amount of rocks. Since, the estimated soil loss using the RUSLE is based on factor values, the result is a direct reflection of these inputs.

Looking at the rainfall records of the area, the summer months have by far the higher share of erosivity as a reflection of the dominant rainy conditions during this time of the year. In addition, the variation of erosivity over the years of the same period is less than that of the relatively dry months. All these properties are extracted from a single station in the reserve and therefore it is impossible to divide the area into various erosivity units. On the assumption that rainfall in the area is correlated with topography, the station might be a good representative of the mid-altitude areas.

Soil erodibility of the area, which is the result of soil physical and organic characteristics, shows that the inherent resistance of the soil is high. The likely reason for this is the dominance of clay and sand contents and the low content of silts.

The topography of the reserve has an undulating nature. A more clear indication of the landscape was represented by the slope that varies considerably, with the valley slopes through which the Bushman's River flows, and the eastern escarpments having most of the steep landscapes. Flow accumulation of the area was used as slope length and this has proved quite interesting in accounting for the direction of water movement. Slope and flow accumulation were then combined and the results further elaborated the adverse effect of the topography on soil erosion. Nevertheless, these steep areas are the very areas that have high stone cover and shallow soils.

The cover factor estimated based on the USLE correlated well with the MSAVI. Hence, an equation relating the two was applied to create the final cover factor. Obviously, high values of C, which are related to low amount of vegetation cover, are found in the western part and to a much lesser extent in the southern far end of the reserve. This indicates that the bare or sparsely-vegetation-covered soils are exposed to the direct impact of raindrop. On the other hand, the steep areas along the valley of the Bushman's River and the eastern escarpments are among the areas with low values implying good vegetation cover in such areas.

The result of the overall soil erosion should, therefore, be seen as a function of the above factors. Though the important contribution of each factor, it appeared from this study that the extent of the effects varied. High erosion losses observed on the steepest lands despite appreciable vegetation cover in such areas indicate the considerably high effect of topography.

Overall, the model's predictive capability is boosted by the use of remote sensing and GIS. As reported on numerous related studies, the conventional procedure of calculating soil loss at plot level and interpolate/extrapolate to a larger area is overcome by the use of these technologies. In this study while some of the data are interpolated using spatial interpolation, others are extracted from digital elevation models and remotely sensed data. On the other hand the DEM of the area, which was provided with an acceptable level of accuracy was employed in this study to derive hydrological parameters with better confidence. Similarly, remotely sensed imagery, which covered the whole reserve, including the less accessible areas, has proved instrumental when compared with ground measurements.

Finally, this study, which covered a small area, has comparative advantage over other reviewed studies in that soil erosion factors were developed at the best possible detail that the logistics allowed.

6.2 RECOMMENDATIONS

The area, in general, is shown to have high potential for soil erosion. However, from its recent history the reserve is managed well with an objective of improving its ecology. Conservation of soil is also part of such a goal. Following the conclusion in this paper that the risk of soil losses in the area is generally high, it becomes necessary that management intervenes to minimize the problem. Because of the spatial nature of the problem, the risks vary in space and hence the intervention type or extent should be applied accordingly.

For management purposes, the risk of soil losses in WGR is divided into three erosion units (Figure 6.1) based on the amount of annual soil erosion potential. In this classification, nevertheless, it should be noted that due to the contagious raster based output, there are pockets of soil erosion risks grouped in an erosion unit that have values different from that erosion unit, thus, the grouping of erosion units is based on the prevalence of a range of selected erosion loss risks.

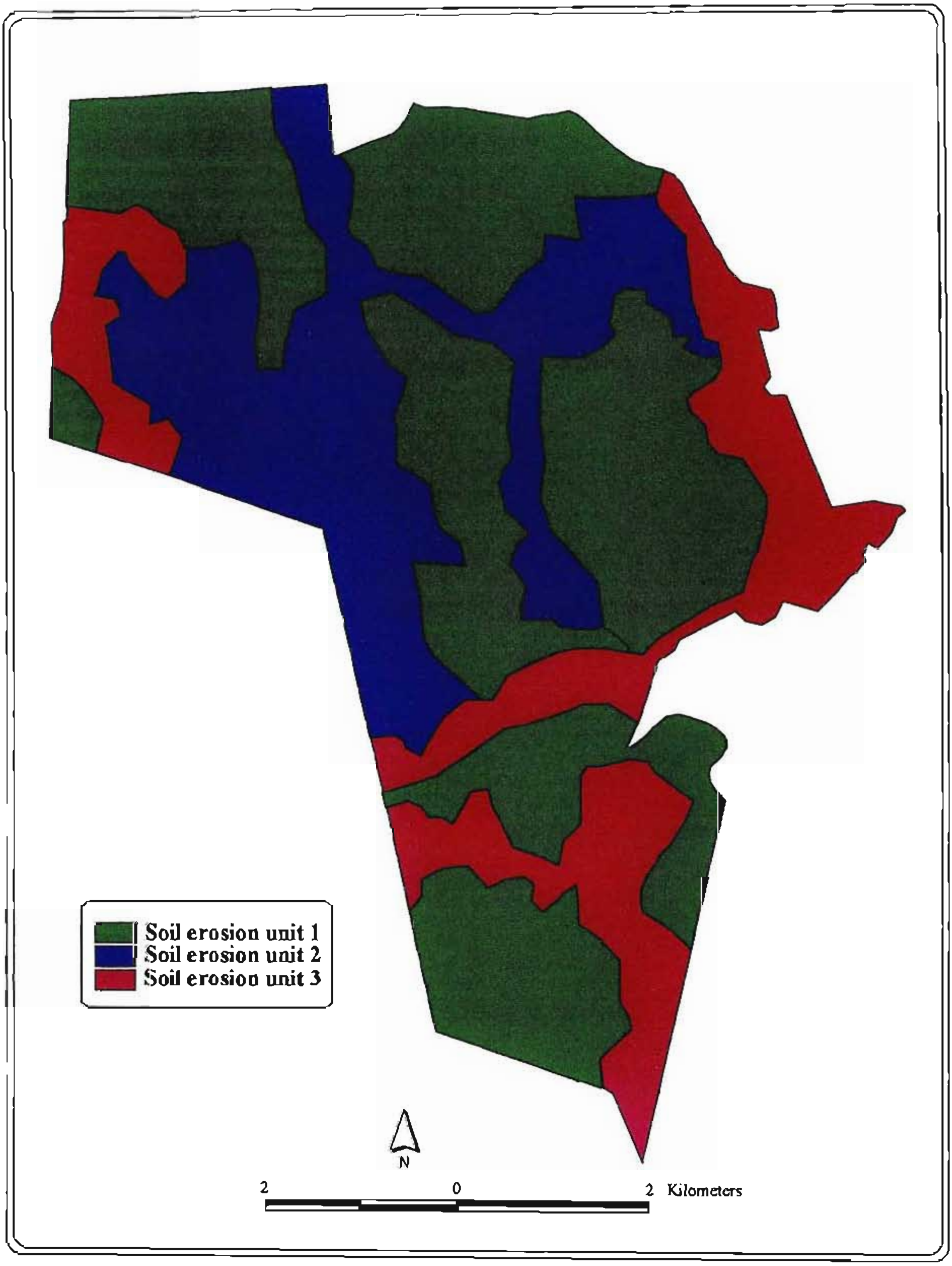

Figure 6.1 *Soil erosion units of WGR*

Soil erosion unit 1

This unit has soil loss risk varying between 0 and 5 ton ha^{-1} $year^{-1}$. The unit covers most of WGR and is found in both the northern and southern parts. Another characteristic that is peculiar to this area is that even risks of soil loss that are accepted to be within the geological limit are also included in it.

This unit is characterized by flat to gently sloping landscape. However, conservation structures are less likely to be a solution for the reason that goes with the service/purpose of the reserve. Following the decision to make the area home to wild animals, the reserve is inhabited by a variety of animals. This has an implication that maintaining physical structures by delineating, and protecting certain parts of the reserve will not be practical. One way of conserving the soil is by increasing the vegetation cover by allowing the natural regeneration, which is said to have shown quite a promising rate of recovery ever since the conversion of the area into a reserve for this purpose. Attaining this could be materialized by limiting the number of animals to the acceptable stocking rate.

Soil erosion unit 2

This unit has a medium soil loss risk that varies form 5 to 20 ton ha^{-1} $year^{-1}$. The areas grouped in this unit are found only in the area north of the Bushman's River. Relatively gentle slopes and less rock covers are characteristics of these areas. Also, these areas have less vegetation cover (tree species) except along streams and watercourses.

In such areas physical structures such as stone bunds are found in several places. Reports have proven that such soil structures have succeeded in reducing soil erosion to a large extent. Many of these structures, due to their age, have been covered by soil and this makes them less effective in reducing water flow down slope. While improving these structures, such approach should be applied in as many areas of this erosion unit as possible. Regarding water streams, it was observed from the field visits that stone packs along watercourses brought about considerable reduction of soil losses. In enhancing the endeavors on this, it is advisable to act on small and developing watercourses since it is easier and more effective to fight at small scale.

Soil erosion unit 3

This unit is the most highly erosion-stricken and susceptible and the potential soil losses in this area vary between 20 and 346 ton ha^{-1} $year^{-1}$. The areas within this unit are found on the steep slopes, which are characterized by comparatively considerable rock or vegetation cover. From field observation, there is little that can be done in these areas in terms of physical structures. Ghebremeskel (2003), in his description of land suitability, reported the difficulty of cultivation on these areas due to their topographic incompatibility.

The best intervention to the areas of this section is to make use of the relatively high vegetation cover, which reduces soil loss by intercepting rain drops, and obstructing surface flow and increasing infiltration rate. Thus, the best way to minimize soil loss in this unit is to maintain the recovery rate. Although most of the areas are dominated by Acacia species that might have a limited effect on interception of raindrops due to narrow leaves of such trees, the southern steep lands have a considerable number of broad-leaved trees. Apart from intercepting raindrops, it was observed that such trees have resulted in a good deal of mulch cover. Therefore, expansion of such species, which is dictated by seed dispersion by birds (Le Roux, 1996), might have a tremendous effect in the future.

The focus on these areas should, therefore, lie on continuing the natural recovery of the vegetation. Camp (2001), based on reports compiled from test sites within the reserve, has indicated that there is tremendous increase in the vegetation community of the area. Though comparisons based on time could not be made, the present organic matter content of the reserve in general is an indicator of the positive move towards successful vegetation revival.

6.3 LIMITATION OF THE STUDY

It can be summarized that this study has achieved its aim by producing maps and quantitative values of soil loss risks in WGR. The use of improved technologies in this study led to the integration of field observations with remotely sensed data with high confidence levels. The final result provides a more comprehensive and tangible insight into the potential soil loss problem of the study area and outlines intervention directives to alleviate the problem. Such study, which can be done with relatively low cost, labour, and time has the potential of applicability elsewhere. This is advantageous in that soil erosion problem varies over space and time and these can be addressed at reasonable scale by applying such methods. With the above-mentioned successes, it is important to note the limitations of the study. Due to the nature of the study, the problems are discussed based on the requirements of each factor in the following manner.

6.3.1 Rainfall

It is generally accepted that spatial variation of rainfall is high in areas where mountainous and rugged topography prevails. Thus, the density of rain gauge stations representing the various rainfall regimes is a necessity. WGR, despite, the above-mentioned topographic nature and non-quantified reports of rainfall dependence on the topography, is represented by one station only. This might not be representative of the whole area. A generally agreed relationship between rainfall and altitude could not be obtained for the study area, either. This implies that the use of the single station to derive an erosivity factor for the whole area is likely to carry over a potential error to areas where the topographic conditions are different.

An additional problem relating to the rainfall erosivity factor is the absence of autographic rain gauge in the reserve. The derivation of this factor in this area based on surrounding stations using regression analysis results in a loss of accuracy though it is assumed that the original EI_{30} is the best representative of this factor.

6.3.2 Erodibility

Erodibility, which was obtained by applying the USLE erodibility nomograph, is a function of various soil characteristics including soil organic matter content. The nomograph applies to soils with organic matter content of not more than 4%. In this study although 11 soil samples were found to have more than 4% organic matter content, it was assumed that the maximum organic matter content was 4%. This implies that soils with organic matter content of higher than 4% were assigned higher K value.

Another limitation in the erodibility factor is its conversion into surface using spatial interpolation techniques. Though the method used in this study is the best in comparison to the other tested ones, it might still remain short of accounting for the possible variability of soil characteristics within short distances. Increasing the number of samples, which was not done in this study for logistical purposes, mainly cost and time, could have minimized this problem.

6.3.3 Conservation practice factor

Field observations and literature indicate that tremendous conservation works were undertaken in the reserve with reported brilliant success in minimizing soil erosion. In the model, therefore, this factor should have been given a value less than 1. Nevertheless, the effects of these works were difficult to estimate as they lack length adequate enough to stretch along contours. Moreover, they are mostly seen in streams as stone packs to check water flow.

6.3.4 Soil loss

Actual rates of soil loss of the reserve may never be known. Natural or simulated rainfall experiments at a selected plot can by no means tell the real amounts of soil loss from an area. One way of generalizing results from plots to a wider space is selecting an average representative condition for the former. This helps to validate the accuracy of results obtained by researches that employ non-plot method. In this study, plot experiment was not undertaken for financial and time constraints. As in many studies, results of other areas with similar environmental and management settings were brought in for comparison.

End note

Based on the set objectives, this study has been accomplished successfully. The study recommends management interventions based on the severity of the risks and the feasibility of executing the recommendations. As a nature reserve with clear purpose of restoring the ecological wealth of the area, management strategies towards achieving this should always, therefore, underlie the conservation of soil in the area. From the point of view of the scale of this study, which was able to deal with the problem in fine detail as the resources allowed, the study has envisaged similar potential applications elsewhere. Nevertheless, such applications should take into account the limitations that have been outlined in this study.

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APPENDIX

WGR soil forms and their brief descriptions as grouped by Hughes (1989).

A-Shallow soils

i) Grey brown topsoils

Mispah (Ms)

This soil form has an orthic A horizon underlain by hard rock (MacVicar, 1991). This soil is found extensively in the reserve and has a depth of less than 300 mm.

Glenrosa (Gs)

This soil form has an orthic A horizon overlying a lithocutanic B horizon that has more than 70% parent bedrock, fresh or partly weathered. This soil is found mainly in the western part of the reserve and has a depth of 300 to 600 mm (Hughes, 1989).

ii) Black topsoils

Mayo (My)

This soil form has a melanic A horizon underlain by a lithocutanic B horizon (Hughes, 1989; MacVicar, 1991). The presence of this soil form is not common in the reserve and apart from what was shown on the map are also found on the steep northern slopes of the Bushman's River valley in association with dolerite boulders (Hughes, 1989).

Milkwood (Mw)

This soil form has a well structured melanic A horizon overlying (MacVicar, 1991). It occurs in the reserve as isolated patches where dolerite is the parent rock (Hughes, 1989).

Immerpan (Im)

This soil form has a melanic A horizon underlain by a hard pan carbonate horizon (MacVicar, 1991). This soil is common only on the steep northern slopes of the Bushman's River valley where dolerite boulders are abundant.

iii) Red topsoil

Shortlands (Sd)

This soil form ha san orthic A horizon overlying a moderately to strongly blocky structured B horizon. It is found through the reserve and is the second most dominant in the reserve characterized by its shallowness except for patches of deeper occurrences (Hughes, 1989).

B-Plinithic horizons

Avalon (Av)

This soil form has an orthic A horizon overlying a structureless yellow-brown apedal B horizon (Hughes, 1989). It occurs mostly on the flatter areas in the western part of the reserve and is relatively deeper than the other forms (Hughes, 1989).

Westleigh (We)

This soil form is similar to the Avalon except that it lacks the yellow-brown apedal B horizon as a result of which it is shallower (Hughes, 1989). The depth range of this soil is between 250-500 mm (Dekker *et al*, 1980).

Glencoe (Gc)

This soil form has an orthic A horizon underlain by a yellow-brown apedal B horizon (Hughes, 1989). It occurs in few areas of the reserve where the shale parent material is impregnated with iron (Hughes, 1989).

C-Duplex soils

This group includes the following soils and are common throughout the reserve. The soil forms of this group have sandier textured topsoils underlain by heavier, strongly structured B horizons making abrupt transition between the horizons (Hughes, 1989).

Estcourt (Es) and Sterkspruit (Ss)

These soil forms have an orthic A horizon overlying a prismaticutanic B horizon that is characterized by its columnar structure (Hughes, 1989).

Swartland (Sw) and Valsrivier (Va)

These soil forms have an orthic A horizon and are underlain by blocky or sub-angular blocky structured B horizon (Hughes, 1989).

D-Black, structured subsoils

Bonheim (Bo)

This soil form has a melanic A horizon and overlies a moderately or strongly structured subsoil. It occurs in the reserve as isolated patches on locally lower-lying topography (Hughes, 1989).

E-Soils of valley sites

Rensburg (Rg)

This soil form has black vertic A horizon (Hughes, 1989).

Dundee (Du)

This soil form has an orthic A horizon overlying recent river deposits (Hughes, 1989).

Katspruit (Ka)

This soil form has an orthic A horizon overlying a gleyed horizon and is the most common of the valley bottom soils (Hughes, 1989).

Oakleaf (Oa)

This soil form has an orthic A horizon overlying a neocutanic B horizon (Hughes, 1989). This soil is found along river banks and on river terraces throughout the reserve and their depth might reach from 400 to 1000 mm (Hughes, 1989).

Champagne (Ch)

This soil form has an organic O horizon where the soil is wet and it always overlies gleyed material. Its occurrence in the reserve is reported only adjacent to the dam that is located near the Campsite (Hughes, 1989).