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Water erosion risk assessment in South Africa: towards a methodological framework

by

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Water erosion risk assessment in South Africa: towards a methodological framework

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ABSTRACT

Soil erosion is a major problem confronting land and water resources in many parts of the world and the spatial extent should be assessed and continually monitored. The combination of existing erosion models and remote sensing techniques within a Geographical Information System framework is commonly utilized for erosion risk assessment. In most countries, however, especially in developing countries such as South Africa, there is still an absence of standardized methodological frameworks that deliver comparable results across large areas as a baseline for regional scale monitoring. Assessment at the regional scale is often problematic due to spatial variability of the factors controlling erosion and the lack of input and validation data. Due to limitations of scale at which techniques can be applied and processes assessed, this study implemented a multi-process and multi-scale approach to support establishment of a methodological framework for South African conditions. The approach includes assessment of (*i*) sheet-rill erosion at a national scale based on the principles and components defined in the (Revised) Universal Soil Loss Equation, (*ii*) gully erosion in a large catchment located in the Eastern Cape Province by integrating eleven important factors into a GIS, and (*iii*) sediment migration for a research catchment near Wartburg in KwaZulu-Natal by means of the Soil and Water Assessment Tool.

Case Study *i* illustrates that 20% (26 million ha) of South African land is classified as having a moderate to severe actual erosion risk (emphasizing sheet-rill erosion) and describes the challenges to be overcome in assessment at this scale. Case Study *ii* identifies severe gully erosion affecting an area of approximately 5 273 ha in the large catchment (Tsitsa valley) of the Eastern Cape Province and highlights gully factors likely to emerge as dominant between continuous gullies and discontinuous gullies. Case Study *iii* illustrates that a cabbage plot in the upper reaches of a research catchment near Wartburg is a significant sediment source, but is counterbalanced by sinks (river channel and farm dams) downstream. Model assumptions affecting outputs in the context of connectivity between sources and sinks are described. The factor-based nature of this multi-process and -scale approach allowed scrutiny of the role of the main factors in contributing to erosion risk. A combination of poor vegetation cover and susceptible parent material-soil associations are confirmed as the overriding factors in South Africa, and not topography and rainfall as frequently determined in the USA and Europe.

A methodological framework with three hierarchical levels is then presented for South Africa. The framework illustrates the most feasible erosion assessment techniques and input datasets for which sufficient spatial information exists, and emphasizes simplicity required for application at a regional scale with proper incorporation of the most important factors. The framework is not interpreted as a single assessment technique but rather as an approach that guides the selection of appropriate techniques and datasets according to the complexity of the erosion processes and scale dependency. It is useful in determining the relative impact of different land use and management scenarios, as well as for comparative purposes under possible future climate change scenarios.

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DECLARATION

I, J.J. Le Roux declare that the thesis, which I hereby submit for the degree PhD in Geography at the University of Pretoria, is my own work and has not been submitted by me for a degree at this or any other tertiary institution.

SIGNATURE:……………………

DATE:……………………………

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Text following text on this page is quoted from:

Lennox CL. 2007. God's undertaker; has science buried God? Gutenburg Press: Malta.

"*The existence of a limit to science is, however, made clear by its inability to answer childlike elementary questions having to do with first and last things – questions such as 'How did everything begin?'; What are we all here for?; What is the point of living?*" "(Sir Peter Medawar).

"Studying all the parts of a watch separately will not necessarily enable you to grasp how the complete watch works as an integrated whole…

There would seem then to be two extremes to be avoided. The first is to see the relationship between science and religion solely in terms of conflict. The second is to see all science as philosophically or theologically neutral…

…The rational intelligibility of the universe, for instance, points to the existence of a Mind that was responsible both for the universe and for our minds. It is for this reason that we are able to do science and to discover the beautiful mathematical structures that underlie the phenomena we can observe…

…It is, therefore, not illogical that one of the major reasons why we have been given minds is not only that we should be able to explore our fascinating universe home, but also that we should be able to understand the Mind that has given us the home…

…In conclusion, I submit that, far from science having buried God, not only do the results of science point towards his existence, but the scientific enterprise itself is validated by His existence."

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CONTENTS

1. INTRODUCTION

Soil erosion is a major challenge confronting land and water resources in many parts of the world and the problem may get worse in the future due to population growth and potential climatic and land use changes (Prosser *et al.*, 2001; Poesen *et al.*, 2003; Kakembo *et al.*, 2009; Tibebe and Bewket, 2010). Although soil erosion is a natural process it is often accelerated by human activities, for example by the clearing of vegetation or overgrazing (Snyman, 1999). Poor farming practices as well as the trend toward agricultural intensification have been considered to be major causes of erosion. Soil formation is a relatively slow process and, therefore, soil is essentially a non-renewable and a limited resource (McPhee and Smithen, 1984). Prolonged erosion causes irreversible soil loss over time, reducing the ecological (e.g. biomass production) and hydrological functions (e.g. filtering capacity) of soil (Hallsworth, 1987). Boardman (2006) states that the cost of food production is increasing in many parts of the world due to erosion and loss of nutrients. Soil erosion not only involves the loss of fertile topsoil and reduction of soil productivity but is also coupled with serious off-site impacts related to increased mobilization of sediment and delivery to rivers. Furthermore, sediments are a carrier for pollutants which are stored by adhesion on their surfaces. Flügel *et al.* (2003) state that eroded soil material leads to sedimentation/siltation of reservoirs, as well as an increase in pollution due to suspended sediment concentrations in streams which affects water use and ecosystem health. Erosion also aggravates water management problems, especially in semi-arid regions such as South Africa (SA) where water scarcity is frequent.

Given the increasing threat to land resources, especially due to population growth and potential climatic changes, it is important to provide information that can help to target policy to focus on the areas of greatest need (Gobin *et al.*, 2003). It is imperative to prevent negative impacts and to remediate affected areas. Before prevention or remediation of soil erosion can be undertaken the spatial extent of the problem should be assessed and continually monitored. Assessment of erosion, however, is complicated by complex physical processes that involve interaction of a large number of spatial and temporal factors, regional differences and scale dependency (De Vente *et al.*, 2007; Vanmaercke *et al.*, 2011; Parsons, 2012). Soil erosion occurs over many spatial scales including the site of impact from a single raindrop to large catchments, as well as over a large variety of timescales such as a single storm to many decades (Stocking and Murnaghan, 2001). Table 1 summarizes the spatial and temporal scales over which the main soil erosion processes occur. Soil erosion assessments can thus be conducted at a variety of scales using a variety of different techniques (see broad categories and examples in Table 2). Although erosion control measures need to be implemented at the field or hillslope scale, allocation of scarce conservation resources and development of policies require erosion assessment at a regional (catchment to national) scale (Vrieling, 2006). Mapping and modelling are therefore key issues to be addressed as baseline for regional scale monitoring (Martinez-Casasnovas, 2003).

Table 1: Description of the spatial and temporal scales at which soil erosion processes occur.

Spatial scale		**Description/size**	**Associated erosion processes**	**Typical associated temporal scale**
Microplot		Area of about 1 m^2	Rainsplash[1] erosion	Seconds
Land facet & runoff plot		An area of homogeneous topography, soil and land management (Van Zyl, 2004); runoff plots are typically rectangular, being about 20 m long and 2 to 3 m wide	Comprises above, sheet[2] and rill[3] erosion	Minutes – daily
Hillslope		Typically extends from upslope/crest areas to a stream channel with varying topography, soil and land management (Van Zyl, 2007)	Comprises all above and gully[4] erosion	Minutes – daily
Regional scale	Catchment	A land surface which contributes water and sediment to any given stream network (Rowntree and Wadeson, 1999), including smaller (sub)catchments (<10 km^2) to a very large catchment (>10 km^2)	Comprises all above and bank[5] erosion, as well as mass movement[5] (sediment[6] storage in sinks may play a large role but is region-specific)	Daily – annual
	National	Refers to countries generally large in extent	Comprises all above	Monthly – annual
	Global	Refers to the whole world, or the combination of several countries including continental scale	Comprises all above	Annual – decadal

1. Rainsplash erosion is the action of raindrops on soil particles by disrupting and transporting soil particles, as well as compacting soil particles that leads to the formation of surface crust and runoff (Mutchler *et al.*, 1994).
2. Sheet erosion involves the detachment and transport of soil particles by rainsplash erosion and transport by shallow overland flow (Lal and Elliot, 1994).
3. Rill erosion is a process in which flow becomes channelled and numerous small channels of several centimetres up to about 30 cm are formed (Bergsma *et al.*, 1996). Sheet and rill erosion normally occur together and it is virtually impossible to assess them separately with modelling and remote sensing techniques at a regional scale.
4. Gully erosion is a process where surface (or subsurface) water concentrates in narrow flow paths and removes the soil resulting in incised channels that are too large to be destroyed by normal tillage operations (Kirkby and Bracken, 2009).
5. Outside scope of text.
6. The term sediment yield is used to refer to the amount of eroded soil (including suspended sediment and bedload) that passes a designated point at the outflow end or outlet of specific area or catchment during a specific time step (thus the cumulative product of all sediment producing processes in a catchment) (De Vente and Poesen, 2005).

Table 2: Broad categories of soil erosion assessment techniques at different scales.

Assessment technique		Description and examples	Typical scales of application
Field measurements		Physical measurement in the field using specific instrumentation such as plots with or without rainfall simulators (e.g. Dong *et al*., 2012)	Microplot to runoff plot
Modelling	Physical	Based on the solution of fundamental physical equations describing the conservation of mass and momentum of streamflow and sediment transport on a hillslope (Merrit *et al*., 2003). Models such as KINEROS are in many cases spatially distributed and event-based in order to estimate the response (loss and/or sediment yield) of the modelled area to single storm events (e.g. Al-Qurashi *et al*., 2008)	Hillslope
Modelling	Conceptual	Lump representative processes over the scale at which outputs are simulated, but incorporate important transfer mechanisms of runoff and sediment generation in their structure to assess soil loss and/or sediment yield (Merritt *et al*., 2003). Several conceptual models draw on MUSLE where sediment yield is computed using surface runoff and peak flow rate together with the widely used USLE factors e.g. SWAT. These models are often continuous simulation models in order to simulate long periods of time with a time step of 1 day (e.g. Srinivasan *et al*., 2010)	Hillslope to catchment
Modelling	Empirical	Based primarily on the analysis of observations with low input data (Merritt *et al*., 2003), these models, especially the (R)USLE, have been widely used across the globe to assess soil loss and/or sediment yield (e.g. Hagos, 2004).	Hillslope to catchment
Modelling	Semi-quantitative	A combination of descriptive and quantitative procedures to provide a semi-quantitative estimate of soil loss and/or sediment yield (De Vente and Poesen, 2005). Although developed for application to hillslopes, (R)USLE and its derivatives have been used this way in many regional scale erosion studies across the globe (e.g. Gobin *et al*., 2003; Lu *et al*., 2003).	Catchment to national
Remote sensing	Airborne	Their lower altitude allows much higher spatial resolutions than satellite based sensors (Smith and Pain, 2009) and have been widely used to map soil erosion features including photogrammetric methods using stereo images (Flügel *et al*., 2003), synthetic aperture radar interferometry (Hochschild and Herold, 2001), airborne laser altimetry and volumetric measurements e.g. LiDAR (Perroy *et al*., 2010)	Hillslope to catchment
Remote sensing	Satellite	In contrary to airborne systems, provide broad coverage and long time series of data (Smith and Pain, 2009). Techniques frequently used include visual interpretation (Dwivedi *et al*., 1997), correlation between spectral reflectance values (Price, 1993), automatic extraction/classification techniques (Servenay and Prat, 2003), change detection methods (Smith *et al*., 2000) and imaging radar instruments (Metternicht and Zinck, 1998). However, spatial resolutions similar to aerial photography are now obtainable (e.g. SPOT 5, IKONOS, Quickbird, WorldView and GeoEye) and subsequently utilized to map soil erosion features such as gullies at a national scale (e.g. Mararakanye and Le Roux, 2011)	Catchment to national
Qualitative or expert-based		Studies that rely heavily on the knowledge and interpretation of experts and that are generally applied in areas with limited spatial data (Gobin *et al*., 2003). GLASOD was the first study whereby the expert judgments of several soil scientists across the globe were collated to produce a world map of human-induced soil degradation (Oldeman *et al*., 1991), whereas LADA is the most recent expert-based project including six pilot countries (Argentina, China, Cuba, Senegal, South Africa and Tunisia) (Wiese, 2011)	National to global

GLASOD - Global Assessment of Human-induced Soil Degradation; KINEROS – Kinematic Runoff and Erosion model; LADA - Land Degradation Assessment in Drylands; LiDAR - Light Detection And Ranging, MUSLE – Modified Universal Soil Loss Equation; (R)USLE – (Revised) Universal Soil Loss Equation; SPOT 5 - *Syste`me Pour l'Observation de la Terre*, SWAT – Soil and Water Assessment Tool.

The combination of existing models and remote sensing techniques within a Geographical Information System (GIS) framework is commonly utilized for erosion risk assessment (Gau, 2008). In Australia, for example, the SOILOSS model modifies the (Revised) Universal Soil Loss Equation (R)USLE (Wischmeier and Smith, 1978; Renard *et al*., 1994) within a GIS framework according to Australian conditions (Lu *et al*., 2003). In the U.S.A. BASINS (Better Assessment Science Integrating Point and Nonpoint Sources) developed by the U.S. Environmental Protection Agency is interfaced within a GIS framework and allows the user to choose different internally coupled models such as SWAT (the Soil and Water Assessment

Tool developed by USDA-ARS) (Arnold *et al*., 1998). BASINS is used by many federal and state agencies to assess water resource and nonpoint source pollution problems for a wide range of scales and environmental conditions (Gassman *et al*., 2007). In Europe two standardized approaches were developed to provide comparable information on the soil erosion problem across large areas in Europe (Baade and Rekolainen, 2006). The first is based on remote sensing techniques and a simplification of the USLE interfaced in a GIS (van der Knijff *et al*., 2000). The second, namely PESERA (Pan-European Soil Erosion Risk Assessment Project) is a physically-based and spatially distributed model capable of national assessment of soil erosion in Europe by combining plant growth, runoff and sediment transport models (Kirkby *et al*., 2004). In most other countries, however, especially in developing countries, there is still an absence of standardized methodological frameworks that deliver comparable results across large areas as a baseline for regional scale monitoring. For example in SA, soil erosion risk assessment has been conducted in different regions at various spatial scales but each region and scale required different techniques and input data (detail provided in Section 2).

Since no study can incorporate the knowledge of all aspects of erosion, it is important to understand to what spatial and temporal degree one needs to capture process dynamics for the purpose of the study and to apply the most appropriate and practical technique (Gao, 2008). Assessment techniques should be adapted and modified to combine sufficient simplicity for application at a regional scale with a proper incorporation of the most important processes (Gobin *et al*., 2003). Van Zyl (2007) suggests that the purpose and requirements of erosion studies be determined by its objective, the dominant erosion processes and the availability of data. A minimum information requirement approach should be followed where the simplest technique is applied that satisfies the study objectives whilst ensuring that the dominant erosion processes and factors are accounted for. Due to the fact that there are limitations to understanding each erosion process and scale at which assessment techniques can be applied, Kirkby *et al*. (1996) and Drake *et al*. (1999) recommend that three hierarchical levels be implemented. The first level allows for the assessment of the spatial distribution of the erosion risk at a relatively broad scale, followed by a second level that allows for more detailed assessment of the erosion risk. Level three assesses changes that occur rapidly at relatively fine spatial and temporal scales. Importantly, assessment techniques and data requirements should increase in complexity with progression from the first to third level (Van Zyl, 2007).

Research problem

The main research problem identified in this study is that there is a lack of practical methodological frameworks to provide a consistent baseline for regional scale monitoring, especially in developing countries such as South Africa (SA). Assessment at the regional scale is often problematic (worldwide in general but certainly in SA) due to spatial variability of the factors controlling erosion and the lack of input and validation data (Lenhart *et al.*, 2005; De Vente and Poesen, 2005). Water erosion is driven by complex physical processes that involve interaction of a large number of spatial and temporal factors, regional differences and scale dependency (De Vente *et al.*, 2007; Vanmaercke *et al.*, 2011; Parsons, 2012). The lack of appropriate/representative data often necessitates application of techniques outside areas and scales of intended use. However, the use of techniques outside of conditions for which it was developed may lead to large errors by either disregarding important erosion factors or overvaluing less important ones. For example, it appears that the inherent erodibility of soil and parent material are the overriding erosion risk factors in SA (Laker, 2004) and not the climate and slope gradient as frequently determined in the USA and Europe (Vanmaercke *et al.*, 2011). In addition, not all erosion types occurring in specific areas are always taken into account. Most regional studies in SA emphasize the sheet and rill aspects of the erosion cycle but exclude gully erosion thus underestimating soil losses in regions where gullies are prominent (Van Zyl, 2007). According to Boardman (2006) and Parsons (2012), gullying and sediment movement are often ignored due to variability at a regional scale.

The above-mentioned problems of spatial heterogeneity and lack of data in SA are coupled with the availability of a wide variety of approaches and techniques that causes measurement variability (Zhang *et al.*, 2002). Laker (2004) states that erosion research methodologies became more diversified over the preceding few decades but the methods used and the results produced are not comparable with each other. These problems hinder successful soil erosion risk assessment and the development of site- and scale-specific control measures to reduce and prevent soil erosion in developing countries such as SA. With the increase of human impacts on the environment, especially agricultural intensification, there is a need to standardize assessment and monitoring methodologies in order to support efficient environmental management strategies (Rubio and Bochet, 1998; Symeonakis and Drake, 2004). Such considerations highlight the need to establish a methodological framework that delivers comparable results across large areas and a baseline for regional scale monitoring in the country.

Aim and objectives

The study aims at establishing a methodological framework using the most feasible erosion assessment techniques and input datasets for which sufficient spatial information exists, emphasizing simplicity required for application at a regional scale with proper incorporation of the most important factors in South Africa (SA). Assessment will be limited to water erosion, as this is considered the most important form of soil erosion at a regional scale in SA (Garland *et al*., 2000). Due to limitations to understanding each erosion process and scale at which assessment techniques can be applied (Drake *et al*., 1999), a multi-process and -scale approach will be implemented by means of three Case Studies assessing the factors controlling: (*i*) sheet-rill erosion at a national scale, (*ii*) gully erosion in a large catchment and (*iii*) sediment migration for a smaller research catchment. These Case Studies will assist in the establishment of framework and provide relevant information on factor dominance and scale issues. The aim will be achieved through meeting the following objectives:

1. Review on the status of the application of technologies to estimate and monitor soil erosion and sediment processes at a regional scale;
2. Water erosion prediction emphasizing <u>sheet and rill erosion</u> at a **national** scale (Case Study *i*);
3. Establishing the factors controlling <u>gully erosion</u> in a **large catchment** (Case Study *ii*);
4. Modelling connectivity aspects in <u>sediment migration</u> for an agricultural **research catchment** (Case Study *iii*); and thus
5. Establishing a methodological framework for water erosion risk assessment in South Africa.

Due to the complexity of erosion processes, regional differences and scale dependency, a single assessment technique will not be feasible (Vrieling, 2006). Several authors state that the selection of assessment techniques should be determined by the objective of the study, the size of the area (scale), the dominant erosion processes and factors, as well as the availability of data (Morgan, 1995; Gobin *et al*., 2003; Merritt *et al*., 2003; Boardman, 2006; Van Zyl, 2007). A distinction should be made between factors that are useful to have and those which are practical to obtain (Warren and Khogali, 1992). Ideally such a framework needs to provide a comprehensive set of guidelines in order to allow evaluation of (at least) the dominant factors that contribute to different processes (Symeonakis and Drake, 2004). In a knowledge gap analysis for erosion risk assessment in SA, Van Zyl (2007) recommends development of a framework which allows the use of different techniques requiring readily available data, including gully erosion models/mapping and the assessment of agriculturally derived sediments. Therefore, the study does not intend to develop new erosion models or remote sensing techniques, but will utilize universally applied techniques and derive input parameter values within a GIS framework. The emphasis herein is on factor dominance as represented by the structure and spatial elements of frequently applied techniques and current datasets. It is envisaged this framework for water erosion risk assessment in SA will be useful to guide and standardize future regional assessment efforts in the country, including monitoring the effects of land use and climate change on erosion risk.

Project outline

Following the Introduction Section above, Section 2 provides a theoretical background, including a published state of knowledge review. Section 3 presents (in journal paper format) the three Case Studies assessing erosion processes using different techniques at different scales including: (*i*) sheet and rill erosion indicators in SA at a national scale; (*ii*) factors controlling gully erosion in a large catchment; and (*iii*) modelling sediment migration for an agricultural research catchment. Where applicable in the thesis, the text remains the same as that published, but has been reformatted for consistency of style. Given that the Figures are specific to the papers, a detailed list is not provided in the Contents Section. The three Case Studies support the establishment of the methodological framework in Section 4, providing relevant information and scale issues on the main contributing factors. Finally, a summary concludes the study in Section 5. Since Section 2 and 3 comprise of published papers, for completeness the references are included at the end of each section or paper.

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2. THEORETICAL BACKGROUND

Preface

Section 2 comprises one chapter as follows:

Le Roux JJ, Newby TS, Sumner PD. 2007. Monitoring soil erosion in South Africa at a regional scale: Review and recommendations. *South African Journal of Science* **103**: 329-335.

This section provides a state of knowledge review of approaches and techniques used to assess water erosion at a regional scale, including reference to some examples. In a comparative context, the review paper discusses available technologies that are recognized internationally and the techniques and approaches used in South Africa (SA). Since this chapter was published in 2007 it excludes reference to subsequent literature. More recent studies are listed in Table 2 of Section 1 and receive attention in the following sections. The review also provides a discussion of the major assessment-related deficits which have generally remained the same since 2007. These include spatial, temporal and measurement variability in erosion risk assessment studies across the globe, but especially in SA. Furthermore, in contrary to most international studies, previous studies conducted in SA at the regional scale have disregarded important erosion factors and have overvalued less important ones. The review concludes with recommendations for future research, including the need to establish a methodological framework to guide and standardize future regional soil loss monitoring efforts in SA.

The chapter is co-authored with Sumner and Newby. I conceptualized the paper, undertook chapter structure and main text compilation, submission and revision as discussed with co-authors.

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Abstract

Loss of topsoil is one of the major soil degradation problems confronting agriculture throughout South Africa and receives special attention by policy-makers. For effective prevention and remediation, the spatial extent of the problem has to be established and monitored. Recent developments in the application of remote sensing and GIS to the study of soil erosion offer considerable potential in this regard. This paper outlines key technologies available for monitoring, and highlights the problems to be solved at a regional scale. The status of the technologies used in South Africa are reviewed and the more recent studies related to soil erosion presented in a comparative context. Spatial, temporal and measurement variability are major constraints in erosion assessment. Previous erosion studies conducted in South Africa at the regional scale have disregarded important erosion factors and have overvalued less important ones. Different processes and interactions are likely to emerge as dominant when crossing scale boundaries. Such considerations highlight the need to establish a methodological framework to guide and standardize future regional soil loss monitoring efforts.

Introduction

Soil erosion is a major problem confronting land resources throughout the Republic of South Africa (SA). Previous research indicates that over 70% of the country's surface has been affected by varying intensities and types of soil erosion (Pretorius, 1998; Garland *et al*., 2000). Although erosion is a natural process, it is accelerated by human activities such as clearing vegetation or overgrazing (Snyman, 1999). Land degradation caused by soil erosion not only involves the loss of fertile topsoil and reduction of soil productivity, but also leads to sedimentation of reservoirs and increases suspended sediment concentrations in streams with consequent effects on ecosystem health (Flügel *et al*., 2003).

Erosion is a process of detachment and transportation of soil materials by wind or water (Morgan, 1995). Since water is the dominant agent causing erosion in SA (Laker, 2004), it is the focus of this review. Water erosion can occur through rainsplash, in unconcentrated flow as sheet erosion, or in concentrated flow as rill and/or gully erosion (SARCCUS, 1981). Outcomes depend on the combined and interactive effects of erosion factors, namely, rainfall erosivity, soil erodibility, slope steepness and slope length, crop management, and support practice (Wischmeier and Smith, 1978). Assessment of erosion thus requires knowledge of how these parameters change across different scales of space and time. More detail on the factors governing erosion, specifically in a South African context, is provided by Laker (2004), Mulibana (2001), D'Huyvetter (1985) and Garland *et al*. (2000).

Remediation and prevention require that the spatial extent of erosion be established. Many observations of soil erosion have been carried out in SA (Rowntree, 1988; Stern, 1990; Snyman, 1999), but the derived statistical relationships from individual erosion measurements are confined to local conditions and do not provide a sufficiently broad range of input data for regional soil loss monitoring (Vrieling, 2006). Although erosion control measures need to be implemented at the field or hillslope scale, allocation of scarce conservation resources and development of policies demands regional scale assessment (Vrieling, 2006). Geographic Information Systems (GIS) and remote sensing techniques, as well as soil erosion models applied within a spatial context, play an important role at the regional scale. We review available technologies with international standing for this purpose, and the techniques and approaches used in SA. More recent techniques and products related to soil erosion at a national scale receive special attention. The review is followed by a discussion of the major assessment-related deficits and recommendations for future research.

Technologies available for monitoring

A wide variety of techniques are available for assessing soil erosion risk across a wide range of scales (Morgan, 1995; Garen *et al*., 1999; Jetten *et al*., 1999; Smith, 1999; Merrit *et al*., 2003; Aksoy and Kavvas, 2005; King *et al*., 2005; Stroosnijder, 2005; Vrieling, 2006). Slope-scale measurements include field rainfall simulation studies and the use of delineated runoff plots (McPhee *et al*., 1983; Snyman and Van Rensburg, 1986; Stern, 1990; Russell, 1995; Rapp, 1998), which provide valuable data on erosion rates of different crop covers and soil types. Although essential for calibration and verification of soil loss models, such field experiments only apply to one or a few hillslopes and cannot be directly extrapolated to evaluate and monitor erosion for a whole catchment (Sivapalan, 2003). Thus methods designed to analyze and interpret broader spatial scales are becoming increasingly important (EEA, 2003). The advent of recent developments in the application of GIS and remote sensing technology offer considerable potential for meeting these requirements.

Remote sensing

Remote sensing techniques using aerial photographs and satellite remote sensing data have greatly increased the capacity to record and monitor land degradation at the regional level

(Kumar *et al.*, 1996). Important sensor development has taken place through airborne systems including photogrammetric methods using stereo images (Kakembo, 1997; Flügel *et al.*, 2003), synthetic aperture radar interferometry (Hochschild and Herold, 2001), airborne laser altimetry (Ritchie, 2000) and hyperspectral remote sensing (Vrieling, 2006). Although airborne systems and methodologies are useful in the direct identification of erosion, they are not feasible for monitoring erosion at a national scale for which satellite imagery is better adapted.

Five types of satellite-based observations can be undertaken (Stroosnijder, 2005; Vrieling, 2006). Firstly, large eroded surfaces can be visually interpreted, based on deviating spectral properties (Kumar *et al.*, 1996). Secondly, modifications of the former technique involve automatic extraction, including unsupervised and supervised classification, using principal component analysis and the maximum likelihood technique amongst others (Floras and Sgouras, 1999; Servenay and Prat, 2003). Highest accuracy can be achieved using a combination of images from different sensors, e.g., Landsat Thematic Mapper (TM) and Japanese Earth Resources Satellite synthetic aperture radar (SAR) data (Metternicht and Zinck, 1998). Thirdly, direct correlation between erosion and spectral reflectance values sometimes permits the detection of erosion and its intensity. Assuming a relation between vegetation cover and erosion, an empirical relation between erosion and reflection can be used (Price, 1993). The fourth category includes visual interpretation and detection of off-site impacts, such as sediment deposition (Jain *et al.*, 2002) as well as dissolved sediment (Ritchie and Cooper, 1991). The fifth application uses repeat pass SAR interferometry that allows assessment of the change in erosion (Massonet and Feigl, 1998).

Until recently, detection of erosion features with satellite data was difficult due to inadequate resolution (Hochschild *et al.*, 2003). Usually higher resolution data (e.g. *Syste`me Pour l'Observation de la Terre*; SPOT) are better for classifying eroded areas, whereas a larger number of spectral bands (e.g. Landsat TM) results in a better classification of vegetational attributes (Dwivedi *et al.*, 1997). With advances in sensor technology, space-borne data with improved spectral, spatial and temporal resolution is now available. Although not yet reported in the literature, new high resolution satellite imagery such as SPOT 5, IKONOS and Quickbird are very promising for identifying erosion features, such as individual gullies (Lindemann and Pretorius, 1995). However, automatic retrieval of individual features is not currently available due to the heterogeneity of the object itself as well as the environment (King *et al.*, 2005). Most remote sensing studies of soil erosion thus concentrate on the assessment of erosion risk factors, notably, vegetal attributes and, to a lesser extent, soil erodibility, topography and conservation practices (Garen *et al.*, 1999; Vrieling, 2006).

Spatial modelling/analysis

Differentiation between classes of models usually rests on the level of complexity used to represent the soil erosion processes and on the spatial and temporal resolution of the model. Models fall into three main categories: empirical, conceptual and physically-based models (Merritt *et al*., 2003). Table 1 summarises selected models in terms of their classification and scale of application. The best known and widely implemented empirical models for estimating soil loss at the regional scale are USLE developed in the 1970s by the United States Department of Agriculture (USDA), and its upgraded version RUSLE. Although developed for application to small hillslopes, (R)USLE and its derivatives have been incorporated into many regional scale erosion studies across the globe. The European Environment Agency (EEA, 1995), the USDA (NRI, 2001), and the National Land and Water Resources Audit of Australia (Rosewell, 1993; Lu *et al*., 2003), have presented some of the most sophisticated work, namely, CORINE, USLE, and SOILOSS respectively. Conceptual models better represent reality by incorporating the underlying transfer mechanisms of sediment and runoff generation in their structure, representing flow paths in a catchment as a series of storages (Merritt *et al*., 2003). Physically-based models have a much more sophisticated model structure being based on the solution of fundamental physical equations describing streamflow and sediment on a hillslope or in a catchment.

Other categories include continuous simulation models (e.g. SWAT, AGNPS, ACRU), event-based models (e.g. KINEROS, LISEM), lumped models (e.g. RUSLE, SLEMSA) and distributed models (e.g. KINEROS). The first simulates long time periods with a time step of 1 h – 1 day; the second uses a small time step (< 1 min) to simulate a single event; the third employs single values of input parameters with no spatial variability while the last incorporates spatially distributed parameters by taking explicit account of spatial variability.

Table 1: Examples of land degradation approaches and soil erosion models.

Acronym and (model type)	**Name**	**Developed by**	**Aim**	**Time step and partition**
ACRU (Conceptual)	Agricultural Catchment Research Model	Univ. of Natal – Dept. of Agricultural Engineering (Schulze, 1995)	Sub-catchment modelling	Daily Sub-catchment
AGNPS (Conceptual)	Agricultural Non-Point Source Pollution	US Dept. of Agriculture – Agricultural Research Service (Young, 1989)	Estimate runoff water quality from agricultural catchments	Daily Event Cell
CORINE (Empirical and Expert)	Coordination of information on the environment	European Environmental Agency (EEA, 1995)	Soil erosion risk modelling by USLE factor/indicator mapping to target poly actions at a continental scale	Annual Continental, 1:1 million
EUROSEM (Physical)	European Soil Erosion Model	European Union (Morgan *et al.*, 1998)	Compute sediment transport, erosion and deposition throughout a storm	Event Break point Channel Hillslope
GLASOD (Expert)	Global assessment of human-induced soil degradation	International Soil Reference and Information Centre (ISRIC) (Oldeman *et al.*, 1991)	Actual soil erosion based on distributed point data obtained from experts in several countries across the world.	Current risk Global
KINEROS (Physical)	Kinematic Runoff and Erosion model	US Dept. of Agriculture – Agricultural Research Service (Woolhiser *et al.*, 1990)	Event-oriented, physically-based model describing the processes of interception, infiltration, surface runoff and erosion from small agricultural and urban watersheds.	Event Field
LISEM (Physical)	Limburg Soil Erosion Model	Department of Physical Geography at Utrecht University and Soil Physics Division at Winard Staring Centre (De Roo and Jetten, 1999)	Spatially distributed physics-based hydrological and soil erosion model, based on EUROSEM	Event Catchments up to 100 km^2
MEDALUS (Physical)	Mediterranean Desertification and Land Use	European Commission (Kosmas *et al.*, 1999)	To understand and mitigate the effects of desertification in southern Europe	Event, daily Field, catchment
(R)USLE (Empirical)	(Revised) Universal soil loss Equation	US Dept. of Agriculture (Wischmeier and Smith, 1978; Renard *et al.*, 1994)	Lumped empirical models that estimates annual rill and interill erosion based on main soil erosion factors	Annual Hillslope
SLEMSA (Empirical)	Soil loss estimation method for Southern Africa	Department of Agricultural Technical Services (Elwell, 1976)	Lumped empirical model that estimates interill erosion based on main soil erosion factors	Annual Hillslope
SOILOSS (Empirical)	Soiloss: Australian version of the RUSLE	Soil Conservation Service of New South Wales (Rosewell, 1993)	A computer programme that calibrates and modifies RUSLE factors according to Australian conditions	Monthly, Annual Continental Regional
SWAT (Conceptual)	Soil and Water Assessment Tool	US Dept. of Agriculture – Agricultural Research Service (Arnold *et al.*, 1994)	Prediction of the effects of management decisions on water sediment yields for ungauged rural basins	Daily Event Sub-catchment
WEPP (Physical)	Water Erosion Prediction Project	US Dept. of Agriculture – Agricultural Research Service (Nearing *et al.*, 1989)	Soil and water conservation planning and assessment	Breakpoint Continuous Channel Hillslope

The data requirements of models dramatically increase with the introduction of spatial (distributed) and temporal (event-based and continuous time step) complexity. For example, distributed and continuous simulation models require large quantities of spatial and temporal data for weather and land use. Several authors state that the description of water fluxes over and through the soil is the foundation of an erosion model (Garen *et al*., 1999; Jetten *et al*., 1999; Aksoy and Kavvas, 2005). Additional information, in particular changes in soil structure resulting from agricultural activities, greatly improves the quality of results. However, complex models tend to be restricted to research catchments and are prohibitive in terms of the time required for implementation on a regional basis as required by government policies. According to Prosser *et al*. (2001), this is the main reason why empirical models are frequently preferred to more complex models, especially at a regional scale. They can be implemented in areas with limited data and are particularly useful as a first step in identifying sources of sediment.

Furthermore, input errors may increase with increasing model complexity. This prevents the application of American models, such as WEPP and KINEROS, or EU-funded models such as EUROSEM and MEDALUS. According to Garen *et al*. (1999) it is not expected that physically-based models such as WEPP will find use in state and field offices of the Natural Resources Conservation Service (NRCS); formerly the Soil Conservation Service. Instead, the empirical and conceptual models, namely RUSLE, SWAT and AGNPS, were adopted by the NRCS for modelling at the regional scale. A user interface, as developed for the AGNPS and SWAT models, streamlines access to key databases and facilitates the preparation of input data sets in the USA. Techniques involving GIS and algorithms for digital terrain analysis are readily available and are currently improving the hydrologic process description in models. Such algorithms are currently used to identify catchment boundaries, determine stream networks and establish overland flow paths as described by Taudem (Tarboton, 2005), HydroTools (Schäuble, 2003) and Tapes (Wilson and Gallant, 2000).

Soil erosion modelling suffers from a range of problems including data variability, over-parameterization, unrealistic input requirements, unsuitability of model assumptions or misleading parameter values in local context and lack of verification data. Recent assessments of the quality of erosion models showed that, in general, the spatial patterns of erosion are poorly predicted (Jetten *et al*., 2003; Merritt *et al*., 2003). Furthermore, models can rarely be relied upon to give accurate predictions of absolute amounts of erosion. Without adequate input data and calibration, models can only be expected to give a relative ranking of the effects of land management (Garen *et al*., 1999). Input data preparation is a laborious task and the mechanics of operating the models is sometimes complicated (Jetten *et al*., 2003). A large part of the effort goes into the construction of the input data set, often derived from a few basic variables that are available as raw data. Despite these limitations, soil erosion models have been modified and applied to regional scales for scenario analysis, and to make objective comparisons that are important for targeting of research and soil conservation efforts in SA.

Background of erosion assessment in South Africa at a national scale

The Department of Agriculture (DoA) and the Water Research Commission (WRC) funded a number of regional-based research projects in SA. Starting in 1991, national studies are summarised in terms of their methodology and scale of application (Table 2). GLASOD was one of the first major regional scale degradation studies conducted by recognized experts in several countries across the globe (Oldeman *et al.*, 1991), including SA (Laker, 2004). Experts divided soil erosion areas into relatively uniform units based on the most important erosion processes. From this a relative ranking of soil erosion risk per area was obtained and a soil erosion risk map was produced at a continental scale.

Thereafter, the use of remote sensing in monitoring soil erosion on a national scale was investigated in 1993. The Bare Soil Index (BSI) was developed with Landsat TM data, making it possible to detect the status of eroded areas on a national scale (Pretorius and Bezuidenhout, 1994). The BSI proved to be reliable in identifying rural settlements and overgrazed and eroded areas in the Mpumalanga and Eastern Cape provinces. Review of the results indicated, however, that the BSI did not differentiate ploughed fields and sandstone outcrops from eroded areas. Furthermore, due to the limited resolution of Landsat TM data (30m), single gullies and limited rill and sheet erosion could not be delineated.

Most regional-based studies concentrated on the assessment of erosion controlling factors, including, rainfall erosivity, soil erodibility, slope length and steepness, vegetal attributes and conservation practices. These are the well-known USLE erosion factors. USLE (McPhee and Smithen, 1984; Crosby *et al.*, 1986; Smith *et al.*, 1995; Smith *et al.*, 2000), RUSLE (Haarhoff *et al.*, 1994; Smith *et al.*, 1995; Pretorius and Smith, 1998; Smith *et al.*, 2000) and SLEMSA (Schulze, 1979; Hudson, 1987; Smith *et al.*, 2000) have been the most widely applied models in SA. Production of the Erosion Susceptibility Map (ESM) was the first national level attempt to integrate the main erosion risk factors within a GIS framework (Pretorius, 1995). The ESM at a scale of 1:2.5 million was created by integrating spatial data on sediment yield, provided by Rooseboom *et al.* (1992) and Verster (1992), with remotely sensed vegetation data, namely, normalized difference vegetation index from the National Oceanic and Atmospheric Administration – Advanced Very High Resolution Radiometer (NOAA – AVHRR) sensor. A second attempt to integrate the main erosion contributing factors at a national level followed in 1998 with the production of the Predicted Water Erosion Map (PWEM) of SA (Pretorius, 1998). Improvements on ESM involved, inter alia, the inclusion of long-term rainfall erosivity data obtained from the iso-erodent map of Smithen and Schulze (Smithen and Schulze, 1982). Also at a scale of 1:2.5 million, PWEM indicates that a very large percentage of the Limpopo (60%) and Eastern Cape (56%) provinces are under severe threat of erosion, whereas the Gauteng and North-West provinces seem to be the least threatened by water erosion. The methodology of ESM and PWEM, however, was based on a considerable simplification of USLE; by combining soil and slope factors with sediment yield data obtained from Rooseboom *et al*. (1992) and Verster (1992). Since PWEM is only suitable for the prioritization of problem areas on a broad scale, due to the coarse resolution (1.1 km) of NOAA images, research continues at a provincial scale.

Mapping and monitoring of natural resources of the Mpumalanga (Wessels *et al*., 2001a) and Gauteng (Wessels *et al*., 2001b) provinces was completed in 2001 and for the O. R. Tambo and Umkhanyakude ISRDS Nodes, located in northern Eastern Cape and KwaZulu-Natal, in 2004 (Ströhmenger, 2004). Improvements to ESM and PWEM include individual attention to the soil erodibility and topography input factors. Soil erodibility index values were utilized by using SLEMSA. In the absence of soil analytical and experimental data, two alternative sources of soil information were used: soil maps (1:50 000 and 1:250 000) (Soil Survey Staff, 1973–1987) and the Land Type Inventory database (1:50 000) (Land Type Survey Staff, 1972–2006). Topography factors were facilitated by the application of digital elevation models and the unit stream power theory developed by Moore and Burch (1986). Results indicate that areas with high erosion potential occur mostly in subsistence farming areas associated with steep slopes and highly erodible soils. However, some units displayed by the erosion hazard maps gave the wrong impression of current soil loss damage. Erosion rates seem to be over-predicted in some of the subsistence farming areas with steep slopes, as well as in mountainous terrain with long and steep slopes.

The most recent national scale overview was compiled by the South African National Biodiversity Institute (Garland *et al*., 2000). A national soil degradation review was compiled using information obtained from 34 workshops throughout SA during 1997 and 1998. Results were presented as a series of maps illustrating the type and severity of soil degradation of different land use types for each magisterial district of SA. The approach is limited by being lumped for each magisterial district, and due to its dependence on apparently subjective judgments.

Table 2: Summary of erosion assessment projects in South Africa at a national scale (from 1991).

Acronym	Name	Location
GLASOD	Global assessment of human-induced soil degradation	Global (Oldeman *et al.*, 1991) Southern Africa (Laker, 1993)
SDPM	Sediment Delivery Potential Map	Southern Africa (Rooseboom *et al.*, 1992; Verster, 1992)
BSI	Bare Soil Index	National (Pretorius and Bezuidenhout, 1994)
ESM	Erosion Susceptibility Map	National (Pretorius, 1995)
PWEM	Predicted Water Erosion Map	National (Pretorius, 1998)
NRA ISRDS nodes	Natural Resources Auditing Integrated Sustainable Rural Development Strategy nodes	Mpumalanga (Wessels *et al.*, 2001a) Gauteng (Wessels *et al.*, 2001b) OR Tambo and Umkhanyakude (Ströhmenger, 2004)
SANBI land degradation review	South African National Biodiversity Institute land degradation review	National (Garland *et al.*, 2000)
–	Potential and actual water erosion prediction maps for SA	National (Le Roux *et al.*, 2006)
SPS of DoA	Soil Protection Strategy of the Department of Agriculture (Lindemann and Pretorius, 2005)	Tertiary catchments in Limpopo, KwaZulu-Natal and Eastern Cape
–	Sedimentation and Sediment Yield Maps for SA conducted by Stellenbosch University - Department of Civil Engineering and ARC-ISCW	National
NPS Pollution Project	Non Point Source Pollution Project	Mkabela and Berg River research catchments (see Le Roux and Germishuyse, 2007)

ISCW is currently involved in several regional-based erosion studies funded by DoA and WRC (see Table 2). These include: potential and actual water erosion maps of SA, currently being validated (Le Roux *et al.*, 2006); remote sensing (SPOT 5) and modelling (SWAT and RUSLE) of the erosion status of three priority tertiary catchment areas, located in the Eastern Cape, KwaZulu-Natal and Limpopo provinces, identified by the Soil Protection Strategy of the DoA (Lindemann and Pretorius, 2005); sedimentation and sediment yield maps for SA to improve the sediment yield maps of Rooseboom *et al.* (1992) and modelling of runoff and sediment transport processes at field to catchment scale to improve understanding of the requirements and processes accounted for by models with international standing, such as SWAT and KINEROS (see Le Roux and Germishuyse, 2007). The following section discusses how the South African studies compare with the international technologies available for monitoring.

Discussion

Spatial pattern prediction of soil erosion is generally not very accurate due to spatial and temporal variability (Jetten *et al.*, 2003). Although soil erosion has been regarded as an important phenomenon in SA since the turn of the century, one of the weaknesses of South

African soil erosion research is the limited information on where the worst problems are located (Mpumalanga DACE, 2002). Errors are assumed to be high in certain areas because of the unknown input factors, especially the vegetation cover factor for various land use practices. More research is needed to assess the confidence limits for the erosion estimates generated for SA at a national scale.

According to Vrieling (2006), it is striking that many studies across the globe have minimally addressed the issue of validation. Studies merely relate the actual range of quantitative erosion rates to measured or predicted values from literature, and are satisfied when values correlate. This is probably because, other than visual comparison of maps, there are very few pattern comparison techniques (Jetten *et al.*, 2003). According to the EEA (2003), proper validation obtained from applying an erosion model at a national scale is hardly possible. Widespread and long-continued soil loss measurements or observations are limited to selected test areas. In SA, limited plot-scale measurements of erosion (e.g. Cedara Agricultural Research Station in KwaZulu-Natal since 1983) (Russell *et al.*, 1995) allow limited regional validation and calibration of USLE factors. Empirical models still need to be appropriately adapted and validated over a long-term and wide range of conditions in SA.

Soil erosion encompasses a vast array of processes, which makes its assessment difficult to encapsulate in a few simple measures. Erosion occurs over a large variety of timescales such as a single storm to many decades. Furthermore, soil loss occurs over many spatial scales including the site of impact from a single raindrop to large fields and catchments. Therefore, measurements undertaken at one set of scales cannot be compared with measurements at another. In this context, a major limitation of soil erosion assessment is that different processes and interactions are likely to dominate when crossing scale boundaries. Soil erosion processes and parameters important at one scale are frequently not important or predictive at another scale (Wilson and Gallant, 2000). The scale problem is coupled with the availability of a wide variety of approaches and techniques that causes measurement variability. Erosion research methodologies became much diversified during the 1980s and 1990s (Laker, 2004), but the methods used and the results produced are far from comparable to each other. Individual studies have inconsistencies in their definitions and measurement procedures, and usually cover short or irregular research periods. Although monitoring implies multi-temporal sampling, most of the studies mentioned above were confined to the use of field surveys and single date imagery to test the potential of using earth observation remote sensing and GIS as monitoring tools. In this context, there exists no methodological framework or “blueprint” to assess the spatial distribution of soil erosion types at different regional scales in SA.

Regional erosion studies cannot integrate all the erosion factors, but have to incorporate the most important processes. Unfortunately, previous erosion studies conducted in SA at the regional scale disregard important erosion factors. For example, Laker (2004) states that important factors of soil erodibility, such as the parent material, degree of soil weathering and stability against dispersion and crusting, are currently excluded in modelling. Various authors state that geology is probably the most dominant factor controlling the inherent erodibilities of soils in SA (e.g. D'Huyvetter, 1985; Dardis *et al*., 1988; Rowntree, 1998; Laker, 2004). Clay dispersibility is also a key factor and significant research is being conducted to gain an understanding on how it influences erodibility of soils in SA (Stern *et al*., 1991; Bühmann *et al*., 1996). However, erodibility of South African soils and how it affects soil erosion in the country, especially within a spatial context, is as yet poorly understood and needs further investigation.

Several regional studies indicate that the soil erosion risk of SA seems to follow topography poorly and is probably overestimated in some areas with steep terrain (Wessels *et al*., 2001a; Ströhmenger, 2004). Although several studies across the globe demonstrate that soil erosion is very sensitive to the topographical factor of RUSLE (Risse *et al*., 1993; Mitasova *et al*., 1996; Biesemans *et al*., 2000), additional work is still needed to test and validate the suitability of topography indices in SA and how it affects soil erosion in the country.

Another noteworthy regional limitation is that not all erosion types occurring in SA are taken into account. Most erosion prediction models emphasize the interrill and rill aspects of the erosion cycle, but few models predict gully erosion (Bull and Kirkby, 1997; Van Zyl, 2004). This is probably due to the temporal and spatial complexity at which the phenomenon occurs, which is difficult to model; e.g. the importance of paths and cattle tracks in creating gullies (Garland *et al*., 2000; Boardman *et al*., 2003; Hochschild *et al*., 2003). Fortunately, more detailed maps derived from satellite imagery are now available for measuring and monitoring gullies, as well as sheet and rill erosion, on a national scale.

Conclusions and recommendations

South Africa is predisposed to soil erosion due to poor farming practices together with erodible soils. When considered across all land use types, it is clear that soil degradation is perceived as more of a problem in the KwaZulu-Natal, Limpopo and Eastern Cape provinces, and less of a problem in the Free State, Western Cape and Northern Cape. However, our ability to develop cost-effective land management strategies is still limited by sources of error in spatial data, ranging from natural variability to issues of accuracy and precision in mapping techniques. In addition, the spatial problem is coupled with a wide variety of mapping techniques that are equally valid but give different results.

Methodological problems, discussed previously, point to the need to establish a proper framework to guide and standardize future regional soil loss modelling and mapping efforts. Such a framework should outline the different erosion processes and interactions likely to dominate at different scales. In this context, regional modelling should combine the simplicity required for application on a regional scale with a proper incorporation of the most important processes. At the regional scale, it appears that the inherent erodibility of the soil and parent material are the overriding erosion risk factors in SA, and not the slope gradient as determined in the USA.

Furthermore, the framework needs to describe the most feasible erosion assessment techniques, as well as input datasets, for application at different scales. For example, it may be feasible to use qualitative approaches where no model is available that was developed or tested in the region under study. Due to the complexity of erosion processes, regional differences and scale dependency, it cannot be expected that a single standardized operational erosion assessment system will be useful. According to Laker (2004), one should rather adopt a dynamic “evaluation tree” approach which would lead the user through a ranking of factors (e.g. parent material, clay mineralogy) in a specific area.

Finally, further refinement of national erosion assessment will be possible given additional research, including:

- Long-term monitoring of soil erosion (e.g. using field measurement and time-series imagery);
- The production of more accurate erodibility maps at a national scale;
- Monthly erosivity estimations in combination with monthly vegetation data in order to capture seasonal variations in soil erosion;
- Spatial modelling techniques to predict gully erosion extent at national scale;
- The use of high resolution imagery (SPOT 5) to extract erosion features at a national scale;
- Careful calibration and validation of prediction models and model components, especially when applied to large geographical areas.

The advent of new techniques and approaches of erosion assessment and recent developments in the application of GIS and remote sensing techniques offer considerable potential for meeting these requirements.

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3. CASE STUDIES

Preface

Section 3 comprises three chapters as follows:

Le Roux JJ, Morgenthal TL, Malherbe J, Sumner PD, Pretorius DJ. 2008. Water erosion prediction at a national scale for South Africa. *Water SA* **34**(3): 305-314.

Le Roux JJ, Sumner PD. 2011. Factors controlling gully development: Comparing continuous and discontinuous gullies. *Land Degradation and Development*, In press. DOI: 10.1002/ldr.1083.

Le Roux JJ, Sumner PD, Lorentz SA, Germishuyse T. 2012. Connectivity aspects in sediment migration modelling using the Soil and Water Assessment Tool. *Geosciences* **2**(5). In press.

Different remote sensing techniques and models can be implemented in order to identify and describe different soil erosion processes, including soil detachment by sheet-rill – and gully erosion and soil transported out of the catchments composing the sediment yield. Due to limitations of scale at which techniques can be applied and processes assessed, this Section implements a multi-process and -scale approach by means of three Case Studies assessing the factors controlling: (*i*) sheet-rill erosion at a national scale, (*ii*) gully erosion in a large catchment and (*iii*) sediment migration for a smaller research catchment. These Case Studies will assist in the establishment of a framework provided in Section 4, emphasizing the simplicity required for application at a regional scale with proper incorporation of the most important factors contributing to sediment generation and migration, including the most feasible erosion assessment techniques and input datasets for which sufficient spatial information exists in SA. The Case Studies also provide relevant information on factor dominance and scale issues.

Although all three chapters are co-authored, model simulations, data interpretation, calibration and/or verification were undertaken by me, as well as chapter structure and main text compilation, submission and revision. All three chapters are co-authored by Prof. Sumner who contributed as project supervisor by commenting on preceding versions before and after review as well as by making editorial changes. The first chapter is also co-authored by Dr. Morgenthal who assisted in the production of the vegetation cover factor map of South Africa, Mr Malherbe who produced the rainfall erosivity factor map of SA, and Mr Pretorius who funded the initial project, as well as assisted in the verification of the final water erosion prediction map of SA for the Department of Agriculture Forestry and Fisheries. The third chapter is also co-authored with Prof. Lorentz who gave the idea of assessing connectivity aspects in the Mkabela Research Catchment using SWAT and provided data for calibration and verification of the model outputs. Mrs Germishuyse provided assistance during model setup and data preparation.

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Case Study I: Water erosion prediction at a national scale for South Africa

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Abstract

Erosion is a major soil degradation problem in South Africa, confronting both land and water resource management throughout the country. Given the increasing threat of soil erosion, a need to improve techniques of estimating the soil-erosion risk at a national scale was identified by the National Department of Agriculture and forms the basic premise of this study. Principles and components of the Revised Universal Soil Loss Equation are applied here since the model combines sufficient simplicity for application on a national scale with a comprehensive incorporation of the main soil-erosion factors. Indicators of erosion susceptibility of the physical environment, including climate erosivity, soil erodibility and topography were improved over earlier assessments by feeding current available data into advanced algorithms. Two maps are presented: an actual erosion risk distribution, and a potential erosion risk map that excludes the vegetation cover factor. Actual soil-erosion risk, which relates to the current risk of erosion under contemporary vegetation and land use conditions, was accounted for by regression equations between vegetation cover and MODIS-derived spectral index. The area of land with a moderate to severe potential risk is found to total approximately 61 m. ha (50%). Although more than 91 m. (75%) are classified as having only a very low to low actual risk, approximately 26 m. ha (20%) of land is eroded at a rate greater than a soil-loss tolerance of 10 t/ha·yr, showing the potential to target erosion control to problem areas. The Eastern Cape, Limpopo and KwaZulu-Natal Provinces have the highest erosion potential. Comparison of potential and actual erosion risk indicates that over 26 m. ha (>30% of national land) could be subject to high erosion risk without maintenance or careful management of the current vegetation cover and land use. Although the distribution of the actual erosion risk broadly follows that outlined previously, this study provides an advance on previous assessments of erosion; results are validated more comprehensively than before, and show an overall accuracy of 77%. The paper also describes many of the limitations inherent in regional erosion studies.

Keywords: water erosion, national scale, potential risk, actual risk, RUSLE

Introduction

Soil erosion is an important form of land degradation and is among the world's, and South Africa's, most critical environmental issues. Previous research indicates that more than 70% of South Africa (SA) is affected by varying intensities of soil erosion (Garland *et al.*, 2000). Erosion is a process of detachment and transportation of soil materials by wind or water (Morgan, 1995) and although 25% of SA is highly susceptible to wind erosion (Hoffman and

Todd, 2000), water is the dominant agent causing erosion in SA and forms the focus of the study. Water erosion occurs mostly through rain-splash, in un-concentrated flow as sheet erosion, as well as in concentrated flow as rill and/or gully erosion. Outcomes depend on the combined and interactive effects of erosion factors, namely rainfall erosivity, soil erodibility, slope steepness and slope length, crop management, and support practice. More detail on the factors governing erosion, specifically in a South African context, is provided by Laker (2004). Although soil erosion is a natural process, it is often accelerated by human activities such as clearing of vegetation or by overgrazing (Snyman, 1999). Loss of fertile topsoil and reduction of soil productivity is coupled with serious off-site impacts related to increased mobilization of sediment and delivery to rivers. Eroded soil material leads to sedimentation/siltation of reservoirs, as well as an increase in pollution due to suspended sediment concentrations in streams which affects water use and ecosystem health (Flügel *et al.*, 2003). According to the latest State of Environment Report of SA, soil erosion costs an estimated R2 bn. annually including off-site costs for purification of silted dam water (Hoffman and Ashwell 2001; cited in Gibson *et al.*, 2006). Before prevention of soil erosion or remediation can be undertaken, the spatial extent of the problem should be established.

Table 1 provides a summary of regional-based work undertaken on soil erosion in SA since 1990. Although some approaches are based on the collection of distributed field observations and/or sediment data, most of the studies use a combination of remote sensing and modelling techniques. In 1993, the Agricultural Research Council – Institute for Soil, Climate and Water (ARC-ISCW) was contracted by the Department of Agriculture (DoA) to investigate the use of remote sensing and GIS in soil degradation management. As a result, Pretorius (1995) produced the Erosion Susceptibility Map (ESM) at a scale of 1:2.5 million by integrating a green vegetation cover map from NOAA satellite data with the sediment yield map of Southern Africa (Rooseboom, 1992). Research continued in 1998 to produce the Predicted Water Erosion Map (PWEM) at a scale of 1:2.5 million applying the widely used Universal Soil Loss Equation (USLE) within a GIS framework (Pretorius, 1998). Methodology, however, is based on a considerable simplification of the USLE, by grouping some of the erosion factors (soil and slope) as one. Furthermore, ESM and PWEM only provide percentage differences in erosion between regions without presenting absolute values and are only suitable to prioritize problem areas on a broad scale due to the coarse resolution (1.1 km) of NOAA images. Another limitation is that both studies are based on single date imagery to test the potential of using remote sensing and GIS as monitoring tools. However, erosion occurs over a large variety of timescales, such as a single storm to many decades (Jetten *et al.*, 2003) and single date imagery does not account for the long-term average soil loss as required by models such as the USLE. Previous studies not only cover short or irregular research periods, they also have inconsistencies in their definitions and measurement procedures. For example, the GLASOD and SANBI studies (shown in Table 1) are limited by being lumped for large districts, and due to dependence on apparently subjective judgments. According to Gibson (2006; cited in Gibson *et al*., 2006), the patterns of degradation reported in the SANBI study (Garland *et al*., 2000) are applicable only in a relative sense and are difficult to repeat for monitoring purposes. Perhaps the greatest problem with previous regional assessments of erosion is the lack of comparison and validation of estimates with actual soil losses.

In order to improve spatial modelling of erosion in SA, a need was identified by the DoA to revise model components and techniques of estimating soil-erosion risk on a national scale. In this context the aim of this study is to improve the spatial soil-erosion indicators in SA on a national scale, including rainfall erosivity, soil erodibility, topography and vegetation cover to derive potential and actual water erosion prediction maps. This study provides a significant update on previous assessments of erosion by inclusion of improved or new national datasets on rainfall, soils, topography and vegetation cover which were not available until recently. Soil erosion indicators are further improved by feeding current available data into advanced algorithms. Each factor is assessed as model inputs within a GIS framework and model outputs are displayed by means of potential and actual water erosion prediction maps. Comparison of potential and actual erosion is important in policy terms because it indicates those areas which are inherently susceptible to erosion (potential risk), but which are presently protected at least to some extent by vegetation (actual risk) (Gobin *et al*., 2003). Results are also validated more comprehensively than before, followed by a description of the limitations and challenges that must be overcome in soil-erosion assessment on a national scale.

Table 1: Summary table of regional erosion studies since 1990.

Abbre-viation	Name	Developed by	Aim	Area and scale
GLASOD	Global assessment of human-induced soil degradation	International Soil Reference and Information Centre (ISRIC) (Oldeman *et al.*, 1991)	Actual soil erosion based on distributed point data obtained from various experts. Soil-erosion areas were delineated according to their judgment.	Global Expert/subjective delineations
SDPM	Sediment Delivery Potential Map	Water Research Commission (WRC) (Rooseboom *et al.*, 1992)	To provide spatial data on sediment yield by gathering sediment data and relevant geographical information which influences sediment yield values of catchments	Southern Africa Catchments 14 to 60 000 km^2
BSI	Bare Soil Index	Agricultural Research Council – Institute for Soil, Climate and Water (ARC-ISCW) (Pretorius and Bezuidenhout, 1994)	To detect bare soil and the status of extensive eroded areas on a national scale with Landsat Thematic Mapper (TM) data.	South Africa 30 m
ESM	Erosion Susceptibility Sap	Agricultural Research Council – Institute for Soil, Climate and Water (ARC-ISCW) (Pretorius, 1995)	To investigate the use of remote sensing and GIS in soil degradation management by integrating a green vegetation cover map produced from NOAA AVHRR satellite data with the sediment yield map.	South Africa 1:2.5 million
PWEM	Predicted Water Erosion Map	Agricultural Research Council – Institute for Soil, Climate and Water (ARC-ISCW) (Pretorius, 1998)	Map erosion by integrating the main erosion contributing factors of the USLE in a GIS including the rainfall erosivity map of Smithen and Schulze (1982), the sediment yield map and green vegetation cover map to account for rainfall, soil-slope and vegetation factors.	South Africa 1: 2.5 million
NRA	Natural Resources Auditing	Agricultural Research Council – Institute for Soil, Climate and Water (ARC-ISCW) (Wessels *et al.*, 2001a) (Wessels *et al.*, 2001b)	Map erosion by regional application of RUSLE in a GIS. Soil and topography factors were, for the first time, separately facilitated by: Application of digital elevation models with a resolution of 75 m for the topography factor; and Soil maps (Soil Survey Staff, 1973-1987) were used to link erodibility values to corresponding soil series in the Land Type Inventories on a scale of 1:250 000 (Land Type Survey Staff, 1972-2006).	Mpumalanga & Gauteng provinces 1: 250 000
ISRDS nodes	Integrated Sustainable Rural Development Strategy nodes	Agricultural Research Council – Institute for Soil, Climate and Water (ARC-ISCW) (Ströhmenger *et al.*, 2004)	As above	OR Tambo and Umkhanyakude nodes in Eastern Cape and KwaZulu-Natal 1: 250 000
SANBI land degrada-tion review	South African National Biodiversity Institute land degradation review	SANBI (Garland *et al.*, 2000)	A series of maps illustrating the type and severity of soil degradation between different land use types, using qualitative information obtained from 400 extension workers throughout SA during 1997 and 1998.	South Africa Magisterial districts

Model selection

South Africa covers an area of approximately 121 m. ha and to cope with such a large area, analysis must be carried out on a relatively small scale. According to Gobin *et al.* (2003), the availability of input data is probably the most important consideration when selecting an erosion model on the regional or national scale. It would be impractical to use a sophisticated model if sufficient input data are not available. On the regional scale, the only means of running a complex model would be to assume certain variables and model parameters to be constant (Nearing, 1998). Prosser *et al.* (2001) identified this as the dominant reason why most soil-erosion prediction carried out on a regional scale is based on empirical relationships. The most well-known and implemented empirical model for estimating soil loss at the regional scale is the Universal Soil Loss Equation (USLE) (Wischmeier and Smith, 1978) developed in the 1970s by the United States Department of Agriculture (USDA), and its upgraded version the Revised USLE (RUSLE) (Renard *et al.*, 1994). Although developed for application to hill-slopes, the (R)USLE and its derivatives have been incorporated into many regional scale erosion studies across the globe (NRI, 2001; Gobin *et al.*, 2003; Lu *et al.*, 2003). In South Africa, empirical models have also been the most widely applied including the USLE (Crosby *et al.*, 1983; McPhee and Smithen, 1984; Snyman *et al.*, 1986; Smith *et al.*, 1995; Smith *et al.*, 2000), RUSLE (Haarhoff *et al.*, 1994; Pretorius and Smith, 1998) and the Soil Loss Estimation Method of Southern Africa (SLEMSA) developed by Elwell (1976) (Schulze, 1979; Hudson, 1987).

Although (R)USLE was originally developed for sub-slope-scale soil conservation purposes, the model gained acceptance in regional-scale applications for the following reasons (Lu *et al.*, 2003):

- RUSLE distils soil erosion into a set of measurable primary soil-erosion factors that facilitates the input data accessibility over large regions;
- The factor-based nature of RUSLE allows easy analysis of the role of individual factors in contributing to the estimated erosion rate;
- RUSLE has a simple mathematical form facilitating the handling of large datasets using GIS.

Therefore it was decided to base the current study on a simplification of RUSLE, the primary function of which is the estimation of (long-term average annual) sheet and rill erosion by runoff from slopes in specified cropping and management systems. The model groups the influences on erosion into five categories, namely climate, soil profile, relief, vegetation and land use, and land management practices; the equation is (Renard *et al.*, 1994):

$$A = R.K.L.S.C.P$$

where:

A is the spatial average soil loss in t/ha·yr
R is the rainfall runoff erosivity factor in MJ.mm/ha·h·yr
K is the soil erodibility factor in t/ha per unit R
L is the slope length factor
S is the steepness factor
C is the cover management factor
P is the support practice factor

Factor values were estimated from the currently available natural resource data in digital form.

Definitions, methodology and improvements

A water erosion prediction map was determined through processing and creating a series of images that represent the RUSLE components in digital form (GIS) (see Figure 1). The manner in which soil- erosion indicators are classified and improved for South Africa follows.

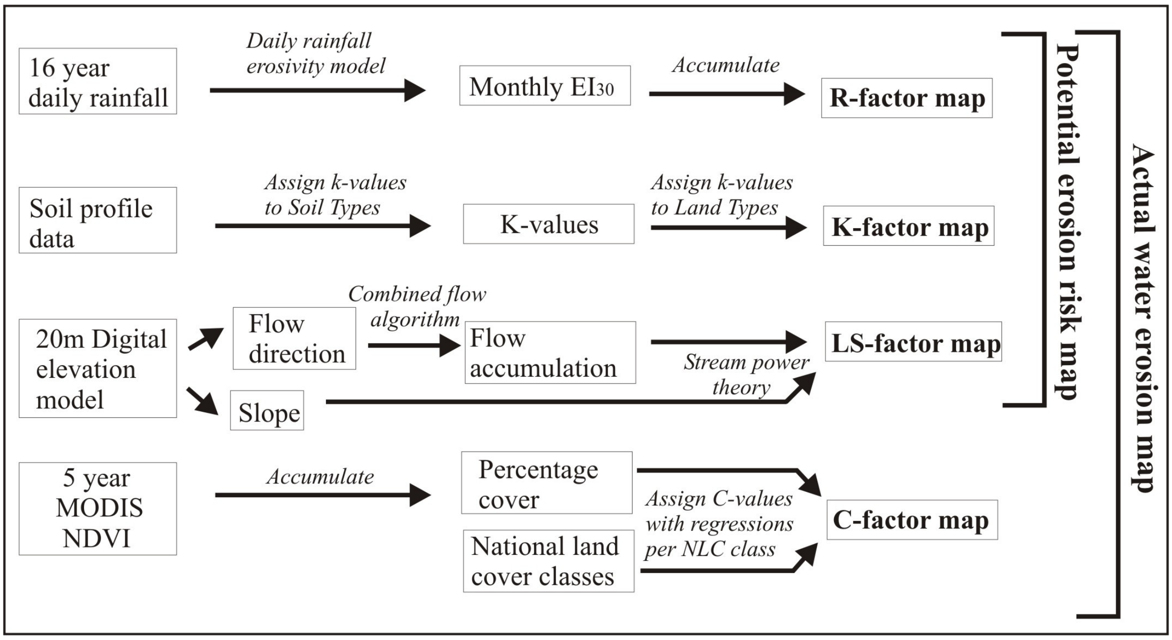

Figure 1: Methodology flow chart for mapping potential and actual water erosion.

Rainfall erosivity (R)

The *R*-factor is the mean annual sum of individual storm EI_{30} values (E is the total storm kinetic energy in MJ/ha/mm and I_{30} is the maximum 30-min rainfall intensity in mm/h). However, reliable and long-term information on rainfall intensity is not available at a regional level and it is necessary to estimate rainfall erosivity from daily rainfall. Here, daily rainfall data (Agrometeorology Staff, 1984-2000) was used as input to the daily rainfall erosivity model developed by Yu and Rosewell (1996a and 1996b) in Australia where it was shown to accurately predict the *R*-factor and its seasonal distribution. Australia has a climate that, similar to SA, ranges spatially between winter rainfall areas in the southwest to a summer rainfall with tropical influences over the northern parts, while large areas over the interior of both countries are classified as semi-arid. Since rainfall is measured at fixed points (weather stations), the inverse distance weight method was used to interpolate data to an EI_{30} surface at 2 km resolution for the entire SA. Using more detailed (stations) and more recent rainfall data than before (e.g. Smithen, 1981) an improved rainfall erosivity algorithm was derived that also compensates for topographical influences.

Soil erodibility (K)

The *K*-factor may be estimated from data on the soil particle size distribution, organic matter content, surface structure and profile permeability using the soil erodibility nomograph (Wischmeier and Smith, 1978). In the absence of soil analytical data in digital form, two alternative sources of soil information were utilised: Soil maps (Soil Survey Staff, 1973-1987) were used to obtain soil erodibility ratings for the individual soil series of the Binomial Soil Classification System of SA (MacVicar *et al*., 1977); and erodibility values were linked to corresponding soil series in the Land Type Inventories (Land Type Survey Staff, 1972-2006) in order to be spatially displayed on a scale of 1:250 000. Using the Soil Loss Estimator of Southern Africa (SLEMSA) model, soil erodibility units were assigned based on an assessment of the surface soil texture, surface soil structure, profile permeability and soil depth of the dominant soils. Subsequently, the SLEMSA erodibility factors were used as a guide to the assignment of RUSLE K-factors (in SI units t/ha per unit R) to all land types of SA. Previously, this methodology was only used at a provincial scale or for smaller areas, including the Mpumalanga and Gauteng Provinces as well as ISRDS nodes (e.g. Wessels *et al.,* 2001a; 2001b; Ströhmenger *et al*., 2004).

Topography factors (LS)

The effects of topography include the effects of slope steepness (*S*) and slope length (*L*). LS-factor maps were extracted from 20 m resolution DEMs (GISCOE, 2001) by means of the widely used stream power equation of Moore and Burch, (1986; Moore and Wilson, 1992). The main difference between this equation and the RUSLE *LS* equation is the use of upslope contributing area in place of flow-path length. The stream power equation is the most widely used method for the extraction of stream networks; to accumulate the contributing area upslope of each pixel through a network of cell-to-cell drainage paths (Band and Moore, 1995; Gallant and Wilson, 2000). Flow-path lines are constructed from flow direction given by an aspect angle. In this study, flow tracing was calculated using a flow algorithm (combined) available in HydroTools (Schäuble, 2003), which is an add-in program for ArcView GIS 3.x. Methodology from previous erosion studies was thus improved by using more detailed digital elevation data (20 m instead of 70 m or higher); and refining the flow tracing using the combined flow algorithm instead of the single flow algorithm used before.

In addition, the soil and slope factors were separately accounted for, instead of grouping them into one, such as in Pretorius (1998).

A potential water erosion map of SA is generated by combining the above indicators, and represents the inherent susceptibility of the soil to rainfall erosion, irrespective of vegetation cover or land use. Actual soil-erosion risk, which relates to the current risk of erosion under present vegetation and land use conditions, was accounted for as follows:

Vegetation cover index (C)

The *C*-factor is the ratio of soil loss from an area with specified cover and management to soil loss from an identical area in tilled continuous fallow. However, since it is not possible to take field measurements at a national scale throughout the year, it was necessary to ascertain how crops change with time by means of remote sensing techniques and other sources of literature (e.g. Acocks, 1988; Low and Rebelo, 1998; National Land Cover, 2000). The widely used NDVI was used in this study as an indicator of vegetation growth determined from images between 2000 and 2004 from the Moderate Resolution Imaging Spectroradiometer (MODIS). MODIS is more advanced than NOAA data previously used with regard to its spatial (250 m^2) and spectral (36 bands) resolution. Subsequently, *C*-values were assigned through regression equations between vegetation cover and MODIS-derived spectral index. The *C*-factor was estimated using the equations based on data from Wischmeier and Smith (1978). Assessment of the support practice factor (*P*) was excluded by setting the *P*-factor to 1. Thus, the estimated soil loss rate for cropping lands reflects erosion rates with no support practices other than cover management. More detail on these procedures is provided by Morgenthal *et al.* (2006) and Le Roux *et al.* (2006). Finally, an actual water erosion prediction map was derived by combining *C*-values with the physical indicators of erosion susceptibility mentioned above.

Results and discussion

Due to the extensive number of input parameters the RUSLE factor maps are provided elsewhere[1] (Le Roux *et al.*, 2006) but the end product of all the input data and erosion factors is presented in the accompanying water erosion prediction maps. Two indicators are proposed as measures of the area affected by erosion: extent to which the total area (e.g. rough estimations per province in million ha) is affected by water erosion, and percentage of

[1] Although not in published paper, all factor maps are provided in Appendix A.

area. Maps are also expressed in quantitative terms and defined into soil loss classes adopted from Bergsma *et al*. (1996) in t/ha·yr: very low (0 to 5); low (5 to 12); moderate (12 to 25); high (25 to -60); very high (60 to -150); and extremely high (>150).

Potential water erosion prediction map

Partially solving the RUSLE equation using climate erosivity, soil erodibility and topography, provides the erosion susceptibility or potential soil-erosion risk of the physical environment. Figure 2 thus represents the worst possible situation, which is the inherent susceptibility of soil to rainfall erosion, irrespective of vegetation cover or land use. The area of land with a moderate to extremely high erosion risk totals approximately 61 m. ha (50%). Figure 2 clearly illustrates that the eastern parts of the country has a much higher erosion potential than the western part of the country. These areas are mostly associated with hill and mountain ranges, regions of cyclonic rain and erodible soils. Conversely, a little over 56 m. ha (46%) of the country is classified as having a low to very low erosion risk, mainly in the Northern Cape (29 m. ha; 13.7%) and North-West Province (7 m. ha; 3.3%) (see Figure 3). Areas of low erosion risk tend to coincide with level plateau areas with low rainfall erosivity.

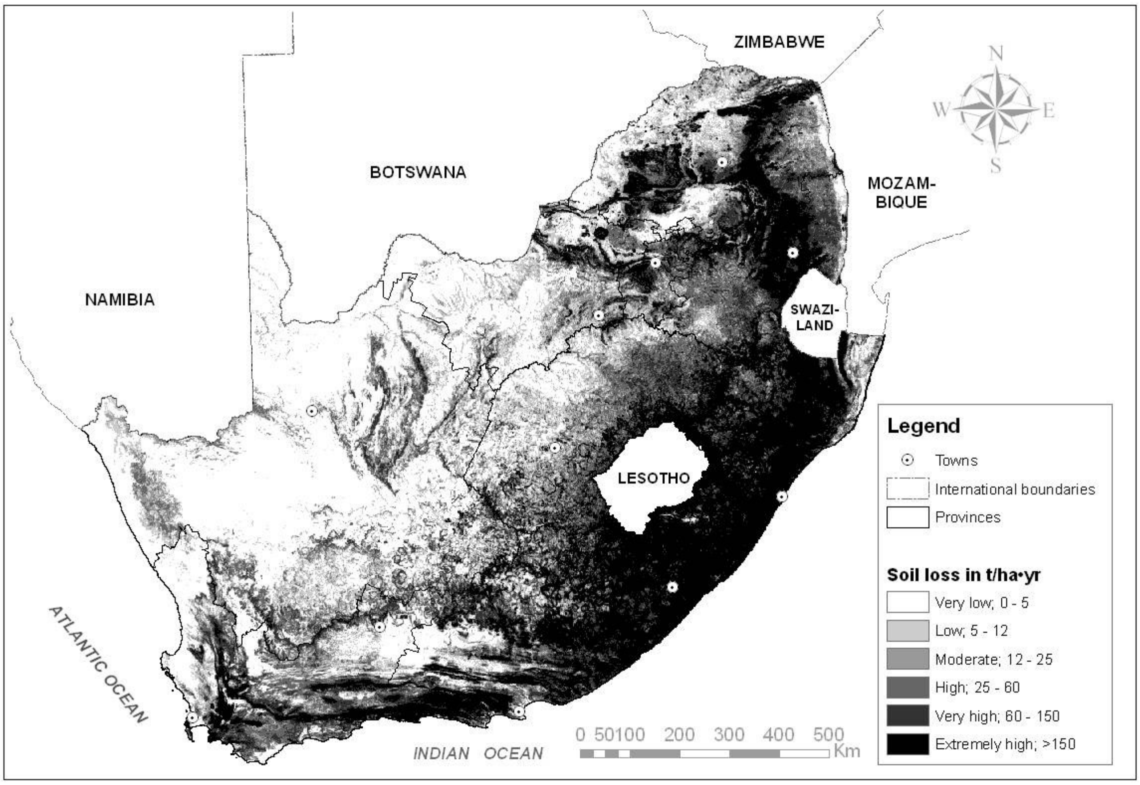

Figure 2: Potential water erosion risk map of South Africa.

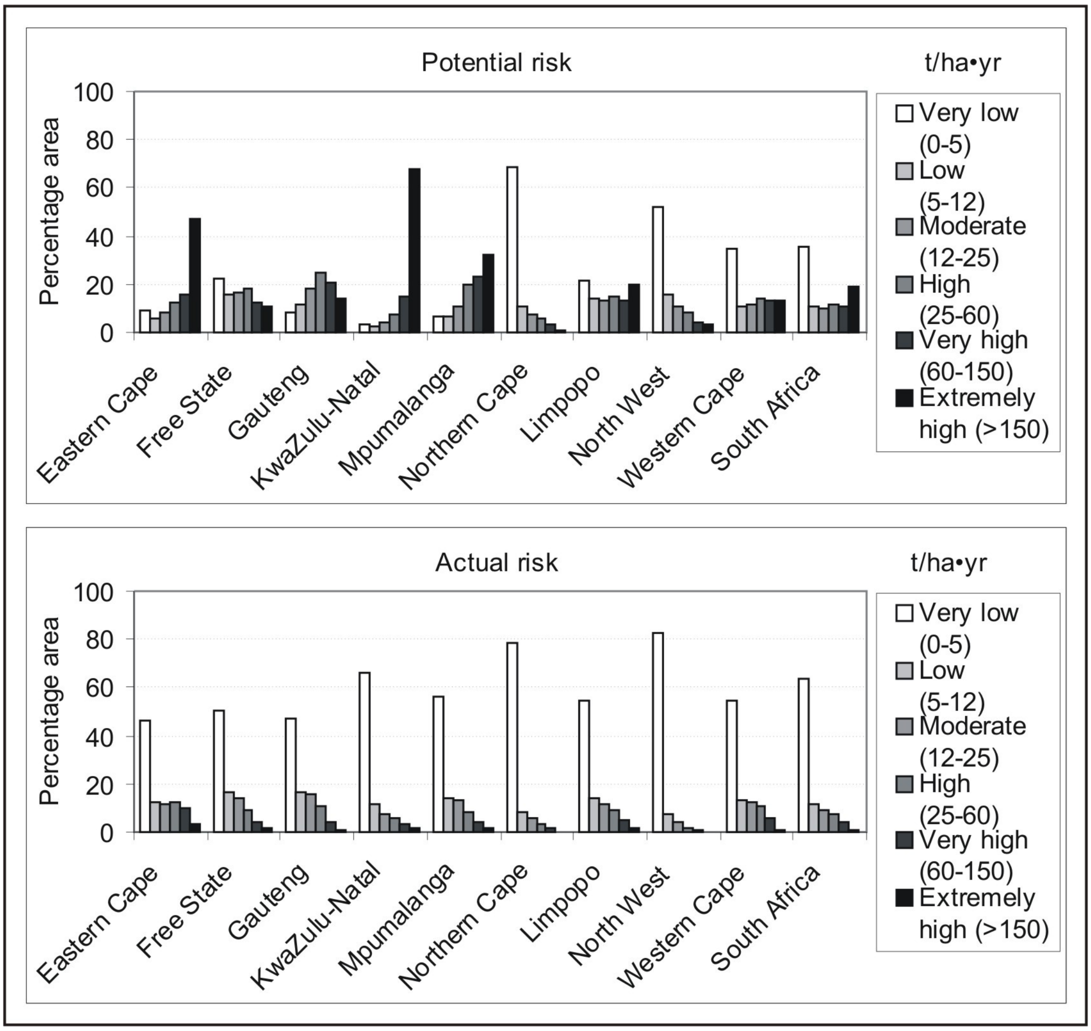

Figure 3: Potential and actual erosion risk of each province expressed as a percentage.

Actual water erosion prediction map

According to the RUSLE, the product of the potential water erosion risk with the cover factor provides the actual water erosion prediction map of SA (see Figure 4). As the data in Figure 3 indicate, the area of land with an extremely high erosion risk totals over 1 million ha (over 1% of the land surface). Although more than 91 m. ha (75%) are classified as having a very low to low risk, approximately 26 million ha (20%) of land is eroded at a rate greater than the suggested soil loss tolerance of 10 t/ha·yr (discussed under validation). In quantitative terms, the average predicted soil loss rate for SA is 12.6 t/ha·yr. It should be stressed that results give a broad overview of the general pattern of the relative differences, rather than providing accurate absolute erosion rates. It is also noteworthy that differences between sediment yield and soil loss can be very high (Garland *et al.*, 2000). Research findings of Scott and Schulze (1991) suggest that soil loss within a catchment can be up to five times greater than sediment yield due to the reduction of the total eroded volume by deposition within the catchment. Consequently, a soil-erosion figure of 12.6 t/ha·yr could correspond with a sediment yield of 2.5 t/ha·yr.

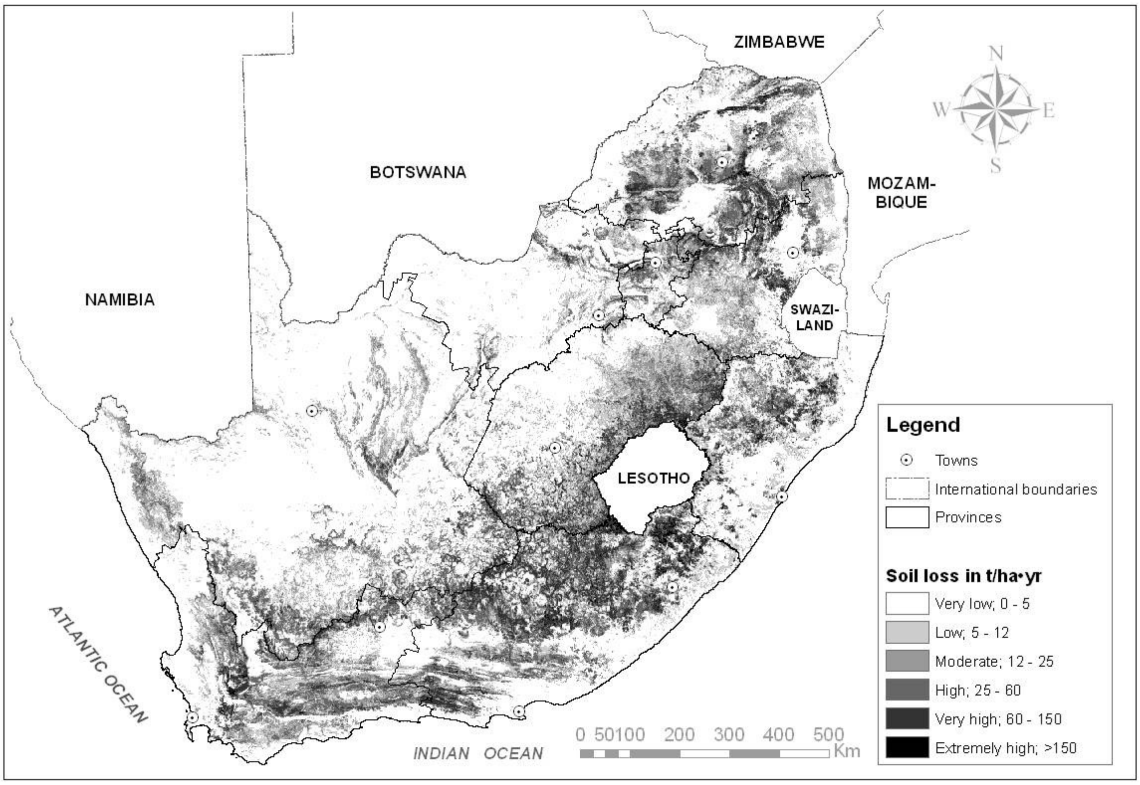

Figure 4: Actual water erosion risk map of South Africa.

Compared to Australia, the average predicted soil loss rate for SA is three times as much than that estimated (4.1 t/ha·yr) by Lu *et al.* (2003). SA has a higher soil loss rate than Australia presumably due to extensive cultivation and overgrazing. A total of 62% of the country is currently under commercial and subsistence farming, including areas that have slopes of 10% or more (National Land Cover, 2000). The areas predicted to be greatly affected by soil loss when compared to the National Land Cover appear to be the degraded unimproved grasslands. Unimproved grasslands are associated with subsistence agriculture where overgrazing of livestock has been excessive. These regions occur widely along the eastern marginal zone, approximately 42 m. ha positioned between the interior plateau and the coast, 0 to 1 200 m a.m.s.l. At the provincial level, the Eastern Cape makes the largest (28%) contribution to soil loss. As is evident from Figure 3, about one third (16 m. ha, 37%) of the province is classified as having moderate to extremely high soil loss.

Comparison between potential and actual water erosion

Comparison of the potential risk with the actual soil-erosion risk indicates those areas which are inherently susceptible to erosion, but which are presently protected by vegetation. It is recognised that there is a huge difference between actual and potential soil erosion, especially along the eastern marginal zone, because low *C*-values (good cover) compensate for the high potential erosion risk. Almost 67% of marginal land has a moderate to severe erosion potential (>12 t/ha·yr), whereas approximately 23% is classified as having a moderate to severe actual erosion risk. Many of these areas are associated with areas of rapid population growth and agricultural intensification, and are thus likely to be at risk. For example, KwaZulu-Natal has large areas of moderate to extremely high potential erosion risk (90%) but relatively low actual erosion risk (18%) due to current vegetation cover. The potential erosion map identifies areas of high soil-erosion potential within some of the natural vegetated areas (e.g. Drakensberg area), but these are natural conditions in steep lands experiencing high intensity rainfall, and do not produce elevated soil-erosion rates. Such comparisons serve to emphasize the importance of vegetation cover for soil-erosion control, and the dangers inherent in changes in land use practice. Over 26 m. ha (at least 30% of national land) would be subject to high erosion risk without maintenance of the current vegetation cover and land use. Importantly, around 4.7 m. ha (37%) of cultivated land surface in SA falls in the high to extremely high potential erosion class. Agricultural intensification could change the land cover, leading to poorer vegetation cover which is the major pressure indicator for soil erosion. The following section compares results with general erosion patterns of erosion risk previously produced.

Comparison with previous studies

Other than visual comparison of maps, there are very few pattern comparison techniques available at a regional scale (Jetten *et al.*, 2003). Only recent regional-scale studies are used for general comparison (see Table 1), since the geographic coverage of field- or plot scale studies is incomplete and cannot provide the comprehensive information required of this study. In general, the distribution of actual erosion risk broadly follows that outlined previously. Very large percentages of the Eastern Cape, Limpopo and KwaZulu-Natal provinces are under severe threat of erosion, whereas Gauteng, the Northern Cape and North-West Provinces seem to be the least threatened by water erosion. The study by Rooseboom *et al.* (1992) of sediment yield is worthy of particular note, as it is based on measurements of fluvial sediment loads and covers a wide geographic area. As with findings here, results indicated that some of the highest sediment yielding areas in SA are situated in the north Eastern Cape and southern Free State, as well as certain areas of KwaZulu-Natal.

It appears that areas of pronounced relief tend to have the highest soil loss rates, including large tracts of the Drakensberg, the former Transkei and Waterberg Plateau. This predicted trend is also consistent with the measurements of Garland *et al*. (2000) who assessed different land use types at a national scale in terms of the main types of soil degradation affecting them. Rill, and gully erosion is the most important types of land degradation on the communal grazing lands of the eastern parts of the country, especially along the escarpment and coastal plain. The study of Pretorius (1998) also indicates that high soil loss rates follow the topography in certain areas with steep terrain, especially along the escarpment.

The predicted results, however, are not in agreement with all the surveys and areas in SA. Disagreements are evident in areas with grazing and subsistence farming on steep slopes. Wessels *et al*. (2001a; 2001b) and Ströhmenger *et al*. (2004) predict high soil loss rates for these areas in Mpumalanga, Gauteng and the OR Tambo and Umkhanyakude ISRDS Nodes located in northern Eastern Cape and KwaZulu-Natal. Current results indicate that not all subsistence farming areas with steep slopes are affected by high erosion rates. Large areas in the OR Tambo node, for example, are not affected by erosion. These regions have a high potential erosion risk but a low actual erosion risk due to good vegetation cover. Current observations indicate that erosion sites occur commonly in subsistence farming areas on soils with high erodibility values. The results of Rooseboom *et al*. (1992) support the concept that areas with erodible soils tend to yield most suspended sediment. Flügel *et al*. (2003) confirm this trend in the Mkomazi catchment in KwaZulu-Natal where erosion sites in informal settlements are mainly located on soils with high erodibility values.

More disagreements are evident in arid areas. Pretorius (1998) predicts much higher erosion rates for the Great Karoo region in the Northern Cape compared to the current study. Possible explanations include the low rainfall and erosivity values for this region, leading to low predicted rates of erosion found here. Although sheet, rill and gully erosion occur commonly in large parts of the Karoo, several of these are relict erosion features. It is postulated that erosion features in some of these areas are of considerable age and may not be contributing to current sediment yields (e.g. Sneeuberg uplands north of Graaff-Reinet) (Boardman *et al*., 2003). Other disagreements are noticeable for the savannah region in northern Limpopo and Northern Cape. Pretorius (1998) predicts a more severe erosion risk for this region compared to the current study. His results may be reasonable since field observations indicate that arid area ground cover is frequently less than its projected vegetation crown cover, which is not always protective against erosion. *C*-values for savannah in northern Limpopo and Northern Cape remain questionable due to the dense tree canopy concealing the poor ground cover when monitored by satellite. Nevertheless, the distribution of the actual erosion risk broadly follows that outlined previously. Such comparisons, however, are not sufficient since the studies differ in their definitions and measurement procedures. By way of validation, the actual water erosion map was compared to data collected during field observations (n = 10 290) including the national Land Type Survey (Land Type Survey Staff, 1972 to 2006) and verification of the National Land Cover (2000) map of SA.

Validation

First, the erosion map was divided into two classes of severity, but not into different erosion types since the soil-erosion maps do not distinguish between erosion types. The two severity classes are expressed in proportion to typical soil loss tolerance values; the maximum rate of soil erosion that can occur and still permit crop productivity to be sustained economically. McPhee and Smithen (1984) proposed a range of soil loss tolerances in SA between 3 t/ha·yr for shallow soils and 10 t/ha·yr for deep alluvial soils. In the current study, areas with very low to low soil loss will have calculated erosion rates close to below the highest possible soil loss tolerance of 10 t/ha·yr. Conversely, areas with moderate to extremely high soil loss will have calculated erosion rates above the soil loss tolerance of 10 t/ha·yr. Second, field observations mentioned above were separated into points where erosion was observed and points where no erosion was observed. In achieving this objective, assumptions were made that all erosion was noted during the surveys and that the current situation is largely unchanged since these surveys in terms of soil erosion. Finally, points where erosion was observed were correlated with areas on the map with moderate to extremely high soil loss values, whereas points where no erosion was observed were correlated with areas on the map with very low to low soil loss values.

In this context, the error matrix shown in Table 2 indicates that the overall accuracy of the actual water erosion prediction map is 77%. For points where no erosion was observed, a distinctly higher number of points (7 168) have very low to low erosion compared to points (1 947) where erosion was observed. For points where erosion was observed, 408 points have very low to low erosion compared to 767 points where erosion was observed. Modellers tend to emphasize the successful part of the simulation only, while more can be learned from difficulties encountered. Therefore, the following section highlights the major constraints of the data and lists several factors that should to be taken into account in such a study.

Table 2: Error matrix between actual erosion map and observation points.

	Erosion	No Erosion	Row Total
n (>10 t/ha·yr) [1]	767	1 947	2 714
n (<10 t/ha·yr) [2]	408	7 168	7 576
Column total	1 175	9 115	10 290
Omission[3]	0.65	0.78	
Commision[4]	0.28	0.94	
Total accuracy	0.77		

1. Number of points on the actual water erosion prediction map that have less than 10 t/ha·yr soil loss
2. Number of points on the actual water erosion prediction map that have more than 10 t/ha·yr soil loss
3. Sample points that have not been correctly classified and have been omitted from category
4. Sample points that have been incorrectly commissioned into another category

Limitations

This study features high levels of spatial and temporal aggregation and incorporation of a relatively small number of casual variables. First, the factors influencing soil erodibility are complex and are influenced by several soil properties. Some of these properties such as organic matter content, stoniness and clay dispersibility were excluded during estimation of the *K*-factor in this study, since the range of descriptive information available for each soil type is limited at a national scale. Laker (2004) states that important factors of soil erodibility, such as the parent material, degree of soil weathering and stability against dispersion and crusting, should not be excluded in modelling. Second, validation of the results indicates that the soil-erosion risk seems to be overestimated for the very steep mountain ranges of the Western Cape and Limpopo Provinces. Although several studies in SA and across the globe demonstrate that soil erosion is very sensitive to the topographical factor of RUSLE (Biesemans *et al*., 2000), additional work is still needed to test and validate the suitability of topography indices in SA and how it affects soil erosion in the country. It appears that the inherent erodibility of the soil and parent material is the overriding erosion risk factor in South Africa, and not the slope gradient, as determined in the US.

Another problem of the regional approach followed is the high variability in space and time of vegetation cover including data such as ground cover, type of land use, and protection measures. For example, *C*-values for Fynbos in the Western Cape are probably too high, leading to over-estimated soil-erosion values. This problem occurs during vegetation senescence when vegetation indices usually decrease even when the cover remains the same (French *et al*., 2000). However, senescent vegetation offers the same protection to the soil as green vegetation and it is important also to detect relatively dry vegetation. Furthermore, this study calculates mean annual erosion, an approach that neglects important seasonal patterns of rainfall erosivity and cover. More specifically, coincidence of erosive rains with low cover in some regions can be a strong control on the mean annual soil loss rates. Finally, the RUSLE-based approach will probably underestimate soil losses in regions where gully and subsurface erosion is prominent (Biesemans *et al*., 2000). These errors, however, can only be challenged at the detailed level (e.g. 1: 10 000 or small catchment scale).

Conclusion and recommendations

This study based soil-erosion prediction on the principles and components defined in RUSLE because it combines sufficient simplicity for application on a national scale with a proper incorporation of the main soil-erosion factors. It also represents a standardised approach and was chosen because of the availability of spatial input data on each of the soil-erosion factors at a national scale. Indicators of erosion, including climate erosivity, soil erodibility, topography and vegetation cover were improved over earlier assessments by feeding current available data into advanced algorithms. Two maps are presented; an actual erosion risk distribution, and a potential erosion risk map that excludes the vegetation cover factor. Comparison of potential and actual erosion is important in policy terms because it indicates those areas which are inherently susceptible to erosion (potential risk), but which are presently protected by vegetation (actual risk).

Large areas of high potential risk occur in KwaZulu-Natal, the Eastern Cape and Mpumalanga, mostly associated with hill and mountain ranges, regions of cyclonic rain and erodible soils. Approximately 50% (61 million ha) of national land has a moderate to severe erosion potential (>12 t/ha·yr), whereas approximately 20% (26 million ha) of land is classified as having a moderate to severe actual erosion risk, exceeding the proposed soil loss tolerance value of 10 t/ha·yr. Comparison of the potential and actual erosion risk indicates that over 26 million ha (30% of national land) would be subject to high erosion risk without maintenance of the current vegetation cover. The Eastern Cape Province makes the largest (28%) contribution to soil loss with approximately one third (16 million ha, 37%) of the province classified as moderate to extremely high.

The distribution of the actual erosion risk broadly follows that outlined previously; high soil loss rates follow the topography in certain areas with steep terrain, especially on the communal grazing lands of the eastern parts of the country along the escarpment and coastal plain. Results, however, are not in agreement with all the previous studies; current results appropriately indicate that not all subsistence farming areas with steep slopes are affected by high erosion rates. Rather, erosion sites occur commonly in subsistence farming areas on soils with high erodibility values. Results are also validated more comprehensively than before, indicating an overall accuracy of 77%. Certain obvious anomalies (e.g. Karoo, Fynbos and savanna regions) reflect the lack of more accurate soil and vegetative cover data for SA. This study features high levels of spatial and temporal aggregation and incorporation of a relatively small number of casual variables. The national-scale information presented here cannot be used to make decisions at a small-scale (farm-scale or on a pixel by pixel level).

Despite these limitations, results remains useful for regional evaluation and serve as an important basis for the determination of areas where soil conservation should be emphasised. Further refinement will be possible given additional research, including:

- The production of more accurate erodibility maps at a national scale by incorporating key factors such as clay dispersibility and parent material;
- Application of RUSLE on a monthly averaged basis by calculating appropriate erosivity and cover factors for each month (in order to capture seasonal variations in soil erosion);
- New high resolution satellite imagery such as *Syste`me Pour l'Observation de la Terre* (SPOT 5) for detecting individual erosion features, especially gully erosion from local to regional scales;
- Establishment of a methodological framework to guide and standardise future regional soil loss modelling and mapping efforts. In conclusion, regional studies should combine the simplicity required for application on a regional scale with a proper incorporation of the most important processes. The development of methods that preserve information across scales or quantify the loss of information with changing scales has become central in erosion studies.

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Case Study II: Factors controlling gully development: Comparing continuous and discontinuous gullies

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Abstract

Gully erosion is a degradation process affecting soils in many parts of the world. Despite the complexity of a series of collective factors across different spatial scales, previous research has not yet explicitly quantified factor dominance between different sized gullies. This factorial analysis quantifies the differences in factor dominance between continuous and discontinuous gullies. First, gullies (totalling 5 273 ha) visible from SPOT 5 imagery were mapped for a catchment (nearly 5 000 km^2) located in the Eastern Cape Province of South Africa. Eleven important factors were integrated into a geographical information system including topographical variables, parent material-soil associations and land use-cover interactions. These were utilized in a zonal approach in order to determine the extent factors differ between continuous and discontinuous gullies. Factors leading to the development of continuous gullies are gentle footslopes in zones of saturation along drainage paths with a large contributing area, erodible duplex soils derived from mudstones, and poor vegetation cover due to overgrazing. Compared to continuous gully conditions, more discontinuous gullies occur on rolling slopes where the surface becomes less frequently saturated with a smaller contributing area, soils are more stable and shallow. Factorial analysis further illustrates that differences in factor dominance between the two groups of gullies is most apparent for soil factors. A combination of overgrazing and susceptible mudstones proves to be key factors that consistently determine the development of continuous and discontinuous gullies.

Key Words: Gully erosion, continuous, discontinuous, factor dominance.

Introduction

Gully erosion is a major soil degradation problem, confronting both land and water resource management in many parts of the world (e.g. Descroix *et al.*, 2008; Kheir *et al.*, 2008; Kakembo *et al.*, 2009). It is a process where surface (or subsurface) water concentrates in narrow flow paths and removes the soil resulting in incised channels that are too large to be destroyed by normal tillage operations (Kirkby and Bracken, 2009). Although gully erosion is a natural process, it is most often triggered or accelerated by human activities such as clearing vegetation and overstocking (Valentin *et al.*, 2005). Once initiated, individual gullies can expand into a network of active gullies that contribute significantly to soil loss in a catchment (e.g. Martinez-Casasnovas *et al.*, 2003). In addition to the loss of arable land, the eroded material leads to sedimentation of reservoirs, as well as lower water tables reducing water available for plant growth or livestock (Kirkby and Bracken, 2009). To prevent these negative impacts and to remediate affected areas (which can be very costly), the spatial extent of the problem and the factors causing it should be established, followed by regional-based erosion control strategies (Poesen *et al*., 2003; Tamene *et al*., 2006).

Most regional studies across the globe emphasize the sheet and rill aspects of the erosion cycle, but few map and/or model gully erosion (e.g. Martinez-Casasnovas, 2003; Vrieling *et al*., 2007). Perspectives on gully factors have typically been obtained from field scale (<10^{-1} km^2) studies and are confined to local conditions (Vrieling, 2006; Ndomba *et al*., 2009). This is probably due to the temporal and spatial complexity at which the phenomenon occurs since several factors contribute to gully development including topographical variables, parent material-soil associations and land use-cover interactions (Valentin *et al*., 2005). Furthermore, gully contributing factors important in a specific area are not necessarily important in other areas (Sonneveld *et al*., 2005). For example, a factorial analysis by Descroix *et al*. (2008) in the subtropical mountain slopes of Western Sierra Madre underline the separation of gullies in two groups. The first group consists of large gullies on gentle slopes with extended contributing/catchment areas where soils are thick and stone-free. The second group constitutes small gullies that occur mainly on hillslopes characterized by steep slopes with thin and stony soils. However, only a qualitative appreciation of the factors influencing their development has been obtained and the factors distinctively controlling small and large gully development remain poorly understood. Differences in factor dominance between large continuous gullies with a branching network that discharges into a stream/river at the base of a slope and small discontinuous that fade out into a depositional zone have not yet been fully resolved.

In this context, the aim of the study is to quantify the differences in factor dominance between continuous gullies (*cgs*) and discontinuous gullies (*dgs*). This will be achieved by accurately mapping gullies in a large catchment (nearly 5 000 km^2) followed by integrating a variety of ancillary information in the form of spatial data layers, also referred to as gully factor maps, into a geographical information system (GIS). A specific catchment located in the Eastern Cape Province of South Africa is used for this purpose coded as tertiary catchment 35 by the South African Department of Water Affairs. The catchment was chosen for its high erosion risk on high potential agricultural land (Le Roux *et al*., 2008a; b). The study highlights gully factors likely to emerge as dominant between *cgs* and *dgs* and provides insight regarding the interplay of eleven important causal factors, collectively disregarded in previous research. The implications of the results are also outlined to assist the design of appropriate strategies targeted at area-specific management of the major causative factors of gully erosion, including the formulation of preventative measures in susceptible areas. Temporal scales are beyond the scope of this research and the study does not distinguish between active and passive gullies.

Site description

The catchment lies between 30º 46' 58'' and 31º 28' 55'' south and 27º 55' 56'' and 29º 13' 47'' east in the Eastern Cape Province of South Africa, north of the town Mthatha (formerly Umtata) (see Figure 1). Elevation ranges from 168 m at the catchment outlet in the southeast to 2 730 m in the Drakensberg mountains. The catchment area of 4 924 km^2 is drained mainly by the Tsitsa River, which flows into the Mzimvubu River after a flow length of approximately 200 km from northwest to southeast. Landforms are complex, ranging from very steep (40%) mountain slopes of the Drakensberg to gently undulating footslopes (2%) and nearly level valley floors. The climate is sub-humid with mean annual rainfall ranging from 672 mm in the lower plains to 1 327 mm in the mountains. Vegetation is largely influenced by altitude, as well as by grazing and burning. The catchment is mainly dominated by grassland including montane, subalpine and alpine belts with pockets of shrub and woodland or Protea savannah (Killick, 1963 as cited in Flügel *et al.*, 2003; Low and Rebelo, 1998). According to the National Land Cover (2000), natural vegetation covers approximately 3 400 km^2 (70%) of the catchment area. The main land use is subsistence grazing (540 km^2 or 11% of the catchment) with minority land uses including forest plantations (4.3%) and commercial agriculture (1.2%). The geology consists of a succession of Beaufort Group sedimentary layers of the Permian Age (Council for Geoscience, 2007). Adelaide mudrock is succeeded by various layers of sedimentary deposits including Tarkastad Mudstones and alternating sandstones of the Molteno, Elliot and Clarens Formations with overlying Drakensberg basaltic lava. Soils from the Tarkastad and Molteno Formations in the central part of the catchment are associated with duplex soils (Land Type Survey Staff, 1972-2008) that are highly erodible with widespread gully erosion evident (Le Roux *et al.*, 2008a).

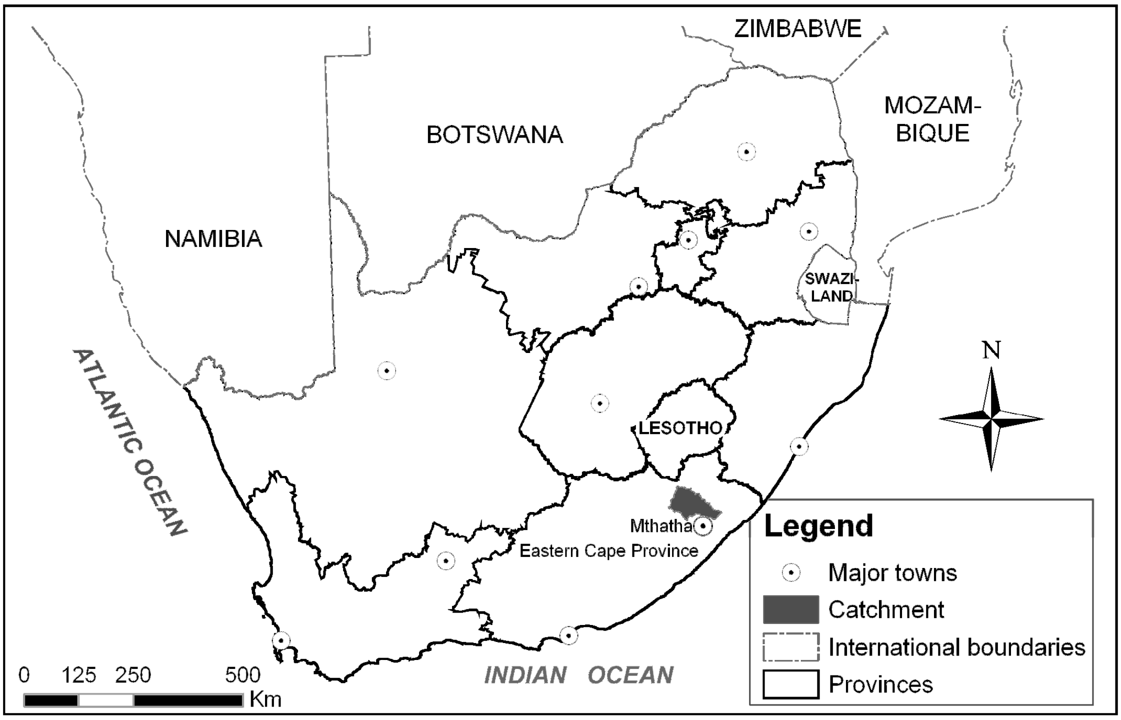

Figure 1: **Location map of study area in the Eastern Cape Province, South Africa.**

Methodology: Gully mapping and factorial analysis

Gully erosion mapping was based on analysis of SPOT 5 imagery from various acquisition dates in 2008. SPOT 5 satellite imagery was utilized because the panchromatic sharpened images at 2.5 m resolution provides high resolution air photo-like quality for gully mapping (Taruvinga, 2008) and was acquired from government agencies for the whole of South Africa. The study resolved to map gully erosion for the whole catchment by means of manual vectorization at a scale of 1:10 000. Although the technique is time-consuming, automated mapping techniques could not express individual gullies with the required accuracy due to their spectral complexity over such a large area. Subsequently, the study effectively distinguished between large continuous gullies (*cgs*) with a branching network that discharges into a stream/river at the base of a slope and relatively small discontinuous gullies (*dgs*) that fade out into a depositional zone.

Several factors contribute to gully development and they have been well described in the literature, including topographical variables (e.g. Desmet *et al.*, 1999; Kheir *et al.*, 2007; Kakembo *et al.*, 2009), parent material-soils interactions (e.g. Laker, 2004; Valentin *et al.*, 2005) and cover management (e.g. Boardman and Foster, 2008; Gutiérrez *et al.*, 2009). The study considered incorporation of rainfall since it is known to be and important driving factor in gully development (Kirkby and Bracken, 2009). Although rainfall varies from 672 mm in the plains to 1 327 mm in the mountains, it was not integrated in this analysis as it does not vary substantially in the central gullied part of the catchment. Since not all gully factors can be taken into account at a regional scale, the study considered incorporation of the most important factors for which regional data already existed, or that could be readily derived for the whole catchment.[2] Descriptions of the gully contributing factors, methods of derivation and data sources are summarized in Table 1. Furthermore, each gully factor layer was categorized into 5 expert-based rankings or classes that, according to observations, uniquely influence gully development. The soil depth factor was categorized into only 3 classes, mainly due to the unavailability of such spatial data (Van Den Berg and Weepener, 2009). These classes allowed assessment of the separate effects of different factors and spatially weighted comparison of environments with unequal surface areas within the catchment, as well as comparison between numerical (S, AS, TWI, LS, K and VC) and non-numerical (TU, GT, LT, SD and LU) data (see Table 1).

A challenge was to assess how factor dominance differs between continuous and discontinuous gullies using the gully factor layers mentioned above. First, an assumption was made that gully factor dominance is associated with the extent of gully erosion within a respective class area. To evaluate differences between these gullies at the large catchment scale, the study postulated that a zonal approach is more appropriate than correlation analyses generally utilized in erosion studies. Multiple regression models, for example, tend to suffer from a limited sample design, subjectivity during factor rating, and a large percentage of variability is usually unexplained (Kheir *et al.*, 2007). Due to the spatially thematic configuration of the gully factor layers it was decided to determine the proportion that each of the above-mentioned 5 classes are affected by continuous and discontinuous gully erosion (by means of zonal functions in the Spatial Analyst extension of ArcGIS 9.3).

[2] Although not in published paper, all factor maps are provided in Appendix B.

Table 1: Description of gully contributing factors and methods of derivation.

Contributing factor	**Description and method of derivation**				
	Class 1 range (area-km^2)	**Class 2** range (area-km^2)	**Class 3** range (area-km^2)	**Class 4** range (area-km^2)	**Class 5** range (area-km^2)
Slope (S)	Gradient (in %) extracted from 20 m resolution DEMs (GISCOE, 2001) using the Deterministic Infinity (D-Inf) multiple flow algorithm in TauDEM (Tarboton, 2004) in ESRI's ArcGIS				
	0-5.00 (1105)	5.00-10.0 (1105)	10.0-19.0 (989)	19.0-34.0 (873)	34.0-100 (852)
Upslope contributing area (AS)	Upslope area per unit width of contour (in m^2) extracted from above-mentioned 20 m resolution DEMs using the D-Inf multiple flow algorithm in TauDEM				
	0-100 (1598)	100-200 (1297)	200-400 (1037)	400-800 (502)	>800 (462)
Topographic wetness index (TWI)	Using TauDEM, zones of saturation is predicted along drainage paths where AS is high and S is low; assuming steady-state and uniform soil conditions (transmissivity) (Wilson and Gallant, 2000)				
	0-0.36 (866)	0.36-0.39 (939)	0.39-0.42 (984)	0.42-0.46 (1039)	0.46-1.00 (1066)
Sediment transport capacity index (LS)	LS is the spatial distribution of soil loss potential that is equivalent to the length-slope factor in the RUSLE where both AS an S is high; assuming the erosion rate is transport limited with uniform rainfall excess runoff (Mitasova *et al.*, 1996).				
	0-1.02 (1110)	1.02-2.30 (1080)	2.30-3.98 (976)	3.98-6.85 (885)	6.85-12.6 (874)
Terrain unit (TU)	Five terrain morphological areas mapped/modelled from a 90 m SRTM DEM (Rodriguez *et al.*, 2005) interpolated to 30 m, using typical topographical algorithms of Evans (1979) and Schmidt *et al.* (2003) in combination with manual vectorization (Van den Berg and Weepener, 2009)				
	Crest (351)	Convex midslope (2284)	Concave midslope (2062)	Footslope (178)	Valley floor (87)
Geology type (GT)	Stratigraphic/lithologic polygon descriptions at a 1:250 000 scale (Council for Geoscience, 2007)				
	Drakensberg basalt, Karoo dolerite (777)	Elliot mudstones, subordinate sandstone (779)	Molteno sandstones (1571)	Alluvium, mudrock, fine-grained sandstone (595)	Tarkastad mudstones (1204)
Land type (LT)	A class of land over which macroclimate, terrain form, and soil pattern each display a marked degree of uniformity at a 1:250 000 scale (Land Type Survey Staff, 1972-2008)				
	Variety of relatively stable soils (304)	Variety of moderately stable soils (1889)	Variety of moderately erodible soils (1063)	Variety of erodible, shallow soils with minimal development (706)	Highly erodible, strongly structured, duplex soils (574)
Soil erodibility factor (K)	Using the SLEMSA model of Elwell (1976), erodibility units were established and used as a guide to the assignment of USLE (Wischmeier and Smith, 1978) K-factors (in SI units t/ha per unit 'erosivity') to land types at a 1:250 000 scale (Le Roux *et al.*, 2008b)				
	0-0.20 (367)	0.20-0.25 (588)	0.25-0.30 (1530)	0.30-0.35 (1564)	0.35-0.70 (871)
Soil depth (SD)	Soil depth was obtained from existing point (753) datasets of the ARC-ISCW, utilized in scripting rules (outside the scope of the text) to create three soil depth class boundaries at a 1:50 000 scale that spatially correlate with land type data (Van den Berg and Weepener, 2009)				
	Shallow (813)	Medium (2140)	Deep (1930)	n.a.	n.a.
Land use (LU)	National Land Cover database of South Africa derived from Landsat TM imagery with a grid cell resolution of 30 m (National Land Cover, 2000)				
	Natural vegetation and plantations (3884)	Urban / Built-up inc. 'townships' (142)	Cultivated, commercial, irrigated and dryland (76)	Cultivated, subsistence, dryland (282)	Degraded unimproved and natural grassland (541)
Vegetation cover (VC)	Fractional vegetation cover (in %) derived from TSAVI on Landsat TM image with a grid cell resolution of 30 m; delivers reliable vegetation cover results for arid and semi-arid grassveld landscapes in South Africa (Flügel *et al.*, 2003)				
	0-20.0 (897)	20.0-30.0 (987)	30.0-40.0 (1115)	40.0-50.0 (928)	50.0-100 (903)

(R)USLE - (Revised) Universal Soil Loss Equation; SLEMSA - Soil Loss Estimator of Southern Africa; SRTM - Shuttle Radar Topography Mission; TauDEM - Terrain Analysis Using Digital Elevation Models; ARC-ISCW - ARC–Institute for Soil, Climate and Water; TSAVI - Transformed Soil Adjusted Vegetation Index.

Results: Gully location map and factor differences

Figure 2 illustrates the spatial distribution of continuous and discontinuous gully erosion in the catchment. Severe gully erosion is identified mainly in the Tsitsa valley located in the central part of the catchment. Table 2 indicates that 4 253 gullies occur in the catchment, directly affecting an area of approximately 5 273 ha (1.1% of the catchment). Only 236 gullies are classified as continuous, yet occupy 2 905 ha (55% of the gullied area). When integrated with drainage networks, gullies reach lengths up to several kilometers and widths up to 100 m. The remaining 4 017 gullies are classified as discontinuous. An error matrix (not shown here) was obtained by comparing the gully vector map with observations in the field (n = 200). In this context, the overall accuracy of the gully map is 93%. Despite the high level of spatial accuracy, however, manual interpretation is incapable of establishing specific erosion process dynamics and spatial information of the driving forces present (Taruvinga, 2008).

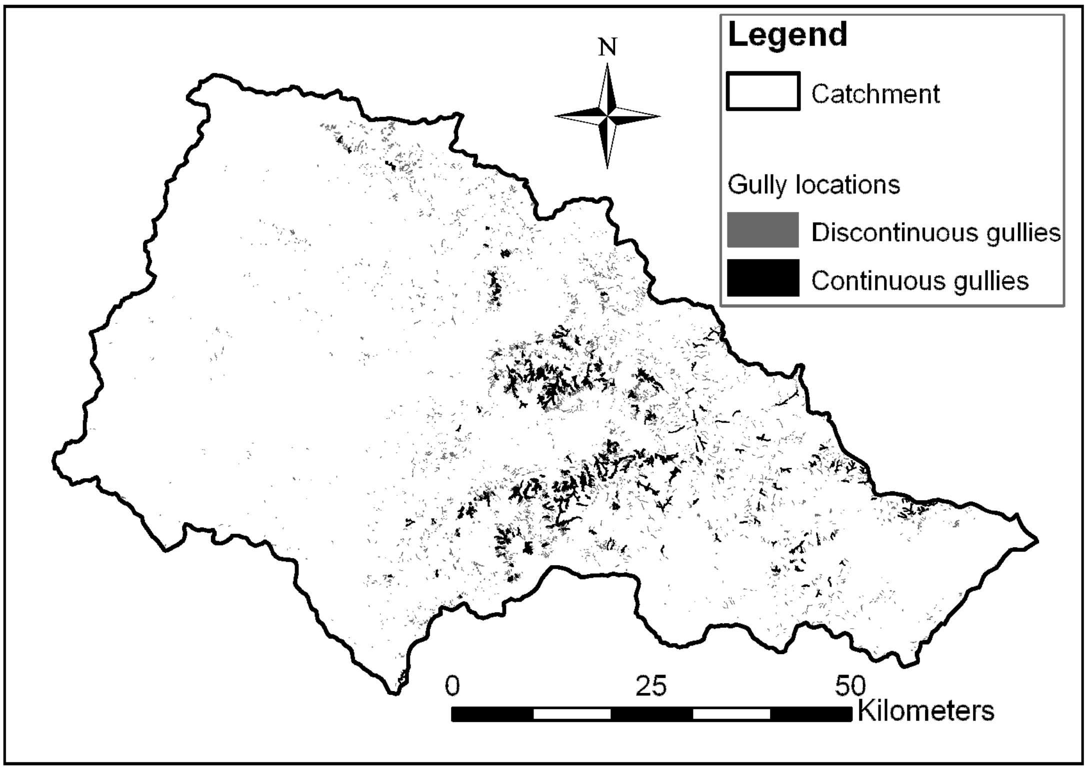

Figure 2: **Gully locations map of the catchment in the Eastern Cape Province, South Africa.**

Table 2: Gully erosion information for the catchment.

Type	Count	Area (ha)	Gullied area (%)
Continuous	236	2 905	55
Discontinuous	4 017	2 368	45

The second category of information is presented as a series of graphs (see Figure 3), expressing the fractions each class (1-5) affected by continuous gullies (*cgs*) and discontinuous gullies (*dgs*). Given that the column height is an indication of gully factor dominance, the most prevalent differences between classes are apparent in Graph-LT, signifying predominant gullying in LT5 (duplex soils). More specifically, approximately 0.0% and 0.1% of LT1 (relatively stable soils) is affected by *cgs* and *dgs* respectively, whereas approximately 5.2% and 1.7% of LT5 is affected by *cgs* and *dgs* respectively. Although not as prominent as LT, the other graphs illustrate similar patterns, with fractions affected by gully erosion gradually and almost linearly increasing or decreasing from classes 1 to 5. Furthermore, results indicate that *cgs* exceed *dgs* in the higher gully classes, whereas *dgs* exceed *cgs* in the lower gully classes (except for Graph-S and Graph-LS). These variations between *cgs* and *dgs* warrant further discussion.

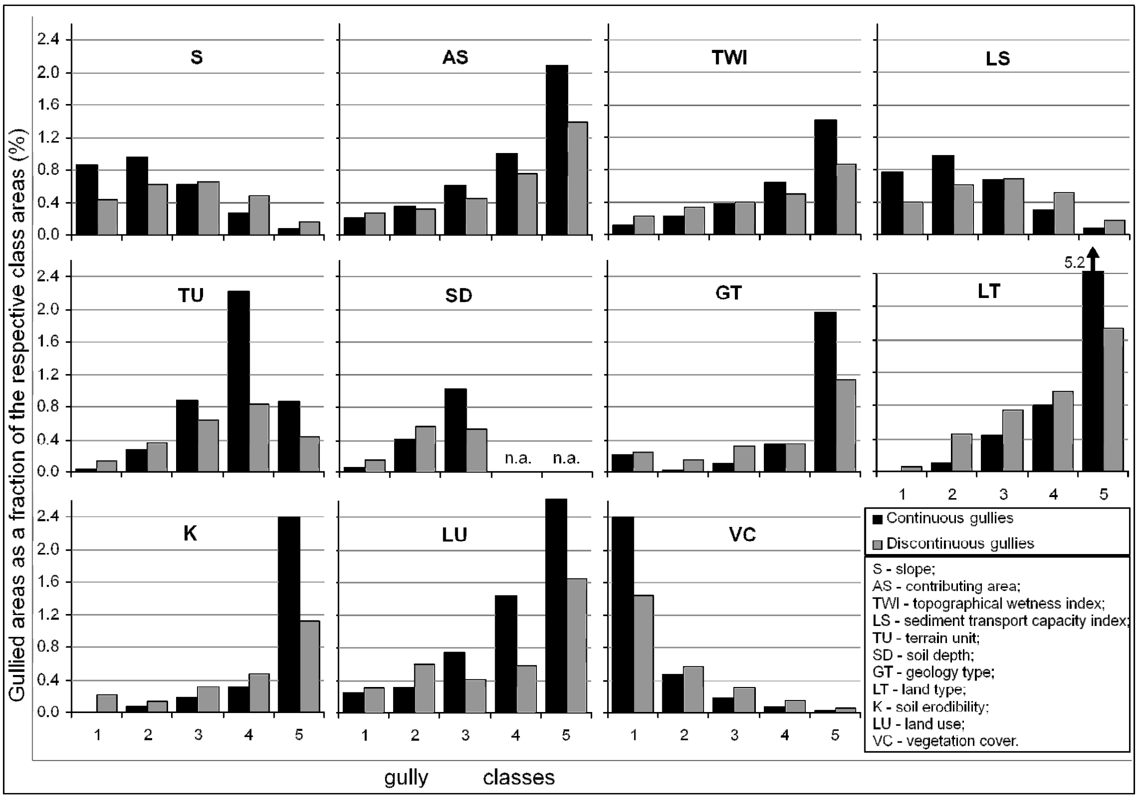

Figure 3: Continuous and discontinuous gullied areas of each class (1-5) as a fraction of the respective class area.

Discussion: Differences between continuous and discontinuous gullies

Foremost, the high variability of gullied areas or fractions within each class is not surprising due to heterogeneity of the landscape. Despite this variability, it is possible to distinguish a hierarchy in causal factors for gully erosion between continuous gullies (*cgs*) and discontinuous gullies (*dgs*). The following discussion describes the gully factors individually but draws some attention to their interdependency. Special attention is given to differences between *cgs* and *dgs*.

Topographical factors

First, gullies in the catchment are mainly located on gentle slopes with gradients less than 10˚ as confirmed in other regions of South Africa (Flügel *et al.*, 2003; Kakembo *et al.*, 2009). Although *cgs* and *dgs* follow a similar trend in this regard, the current study establishes some significant differences. In particular, *cgs* (0.9% of S1) are more prominent on gentle slopes than *dgs* (0.4% of S1), whereas *dgs* (0.2% of S5) are more prominent on steep slopes than *cgs* (0.1% of S5). The reason that *dgs* (the smallest range of gullies) exceed *cgs* (the largest range of gullies) on rolling slopes is coupled with the reason that gully erosion in the catchment is less severe on steep slopes. Tamene *et al.* (2006) found in Ethiopia that gully erosion is less severe on steep slopes, probably due to steep areas being less accessible and less exposed to human and livestock disturbances. Another possible reason is provided by Poesen *et al.* (2003), explaining that the so-called critical drainage area needed for gully initiation decreases as slope steepens. Likewise, Kakembo *et al.* (2009) observed that gullying in several catchments of the Eastern Cape Province predominantly occurs on gentle slopes where the critical drainage area or upslope contributing area (AS) is high.

Upslope contributing area (AS) is an important topographic variable that is frequently linked with gully development. More specifically, gully development largely depends on high AS values (Kheir *et al.*, 2007). Areas with high AS values have high flow accumulation (number of upslope cells that flow into each cell) used to identify drainage areas and flow paths vulnerable to gully erosion (Desmet *et al.*, 1999). It is therefore not surprising that gullies in the catchment are mainly located on areas with a large AS (>200 m^2). It is noteworthy here that, opposite to above-mentioned slope pattern, *cgs* (2.1% of AS5) are more prominent than *dgs* (1.4% of AS5) in areas with large AS values, whereas *dgs* (0.3% of AS1) are more prominent than *cgs* (0.2% of AS1) in areas with low AS values. Differences in AS between *cgs* and *dgs* can be explained by slope length since *dgs* have smaller slope lengths with less flow accumulation/concentration of rain water than *cgs*. Areas with low AS values represent local topographic highs/upper-slopes where flow accumulation required for gully development (especially *cgs*) is limited.

Not surprisingly, areas with high AS values also have high topographical wetness index (TWI) values (areas prone to become wet) and vice versa. Similar to the study of Kheir *et al.* (2007), gully formation in the catchment is particularly favoured in areas with high TWI values (>0.4) representing zones of saturation with high surface soil water along drainage paths where AS is high and slope is low. These saturated areas favour gully formation since the surface soils lose their strength as they become wet. The differences between *cgs* and *dgs* are similar to the above-mentioned AS pattern where *cgs* (1.4% of TWI5) exceed *dgs* (0.9% of TWI5) in areas where TWI is high, whereas *dgs* (0.2% of TWI1) exceed *cgs* (0.1% of TWI1) in areas where TWI is low. Therefore, *dgs* occur more frequently than *cgs* in areas where AS is low and slope is high. Areas with low TWI values represent zones with low surface soil water where gully development (especially *cgs*) is limited.

The sediment transport capacity index (LS) also combines the effects of AS and slope. Areas where LS is high (>4) are vulnerable to erosion due to the generation of sufficient runoff (high AS) with a sufficient level of relief energy (high slope) (Desmet *et al.*, 1999). However, several studies agree that areas with high LS values do not necessarily represent zones where gullies develop (Kheir *et al.*, 2007; Kakembo *et al.*, 2009). Here we confirm that a low proportion of gullied areas in the catchment occur in areas where LS is high. It is noteworthy here that Graph-LS provided in the (Results: Gully Location Map and Factor Differences) Section above appears to be markedly similar to Graph-S, highlighting the distinct predominance of gullies on gentle slopes (as mentioned above). Therefore, for LS, it appears as if slope limits the impact of AS. More specifically, in the catchment more *cgs* (0.8% of LS1) than *dgs* (0.4% of LS1) occur in areas where slope is low, yet AS is high, representing zones of saturation with high surface soil water on footslopes and valley floors. In contrast, more *dgs* (0.2% of LS5) than *cgs* (0.1% of LS5) occur where the slope is relatively high, yet AS is low, representing zones with low surface soil water on topographic highs/upper-slopes.

Several studies in South Africa state that gully development is specially favoured in certain terrain units (TUs), namely footslopes and valley floors (e.g. Descroix *et al.*, 2008; Kakembo *et al.*, 2009). Gully development is favoured in footslopes and valley floors since they represent areas where overland flow is concentrated into preferred pathways of flow (Beckedahl and Dardis, 1988), especially concave hollows adjacent to drainage lines, as opposed to upland convex hillslope sections (Kakembo *et al.*, 2009). The present study indicates that footslopes constitute the preferential gully location zone followed (almost equally) by valley floors and concave midslopes. This pattern is especially noticeable for *cgs* that seems to be expanding from footslopes onto midslopes. More specifically, *cgs* (4.0% of TU3-5) exceed *dgs* (1.9% of TU3-5) in low hillslope and concave sections, whereas *dgs* (0.5% of TU1-2) exceed *cgs* (0.3% of TU1-2) on topographic highs and convex sections. The main reason for this difference is because development of *cgs* is generally restricted to concave areas along drainage paths where soils are deep (whereas *dgs* are not).

Although soil depth (SD) is not a topographical factor *per se*, it is highly correlated with TUs usually increasing downslope or towards the lower hillslope elements (Land Type Survey Staff, 1972–2008). Moreover, gully development also depends on the availability of deep soils (e.g. Descroix *et al.*, 2008; Kakembo *et al.*, 2009). It is therefore not surprising that *cgs* (1.0% of SD3) exceed *dgs* (0.5% of SD3) where soils are deep, whereas *dgs* (0.2% of SD1) exceed *cgs* (0.1% of SD1) where soils are shallow. As a result, relatively large fractions of deep soils are affected by gully erosion, especially where footslopes and valleys are filled with erodible soils derived from mudstones.

Lithological and pedological factors

At the regional scale, several authors note that the inherent erodibility of the parent material (geology type - GT) as the overriding erosion risk factor (e.g. Watson and Ramokgopa, 1997; Tamene *et al.*, 2006). In particular, Laker (2004) indicates that in South Africa various mudstones are susceptible to gully erosion mainly due to highly erodible duplex soils derived there from (soils are further discussed below). Figure 3 above confirms the preferential development of gullies on Tarkastad Mudstones with 2.0% and nearly 1.1% of GT5 affected by *cgs* and *dgs*, respectively. It is noteworthy here that *cgs*, as well as *dgs*, on the other GTs are markedly limited. One would expect to find higher proportions of gullies in GT4 since it contains a combination of transported/unconsolidated alluvium and weak sedimentary mudrock that usually give rise to erodible soils (Laker, 2004). One possibility for this discrepancy is that gully development on GT4 is counteracted by other factors such as good vegetation cover. Another reason for the preferential development of gullies on Tarkastad Mudstones opposed to the other GTs is linked to the soils derived from these mudstones.

Soils from the Tarkastad Mudstones are notably different from all of the other soils investigated in this study. The most prominent feature of these soils (duplex soils) represented by land types (LTs) in class 5, is a permeable horizon overlying an impermeable one. As a result, water infiltrates and saturates the top layer above the impermeable one where it moves along as subsurface flow causing tunnel erosion (Beckedahl, 1998). In addition, these soils are usually dispersive and easily lose aggregation. The tunnel network is exposed as gullies where their roofs collapsed. Here we confirm the preferential development of gullies on duplex soils with 5.2% and 1.7% of LT5 affected by *cgs* and *dgs*, respectively. In contrast, *dgs* (2.2% of LT1-4) are more prominent than *cgs* (1.4% of LT1-4) on a variety of relatively stable red to yellow apedal and litho soils. Evidently, gullied soils do not always, or simply, correlate spatially with weak underlying geology. If so, then Graph-LT (Figure 3) would have reflected the same pattern as Graph-GT. Instead, it seems as if the variability between *cgs* and *dgs* is largely affected by the high spatial heterogeneity of the LTs and the erodibility of their soils.

It is not surprising that extensively gullied LTs have high soil erodibility (K) values (and vice versa). As expected, the K-graph provided in Figure 3 above is markedly similar to the LT-graph. Once more, the distinction can be made between *cgs* (2.4% of K5) being more prominent than *dgs* (1.1% of K5) on highly erodible soils (duplex and dispersive), whereas *dgs* (0.7% of K1-3) are more prevalent than *cgs* (0.3% of K1-3) on a variety of less erodible soils (that weather more slowly with minimal development). As mentioned above, duplex soils are erodible and favour continuous gully development mainly due to the marked increase in clay content from the topsoil to subsoil horizon. As a result, duplex soils have an abrupt transition between the topsoil and the subsoil with respect to texture, structure and consistence (Samadi *et al*., 2005). These soils limit intrinsic permeability since water does not move readily into the subsurface matrix, which leads to increased subsurface flow causing tunnel erosion (Beckedahl, 1998). In addition, several studies agree that soils prone to tunnel erosion are usually dispersive and easily lose aggregation as a result of high sodium absorption (e.g. Rienks *et al*., 2000; Valentin *et al*., 2005). However, due to the lack of spatial information at a regional scale, the correlation between *cgs*, *dgs* and sodic soils still needs further investigation. Collectively, all the factors discussed above highlight areas that are intrinsically susceptible to gully development. The last two factors discussed below are important to highlight areas where gully erosion is extrinsically triggered or accelerated by land use and human-induced reduction of the vegetation cover.

Land use and vegetation cover

As indicated by examples worldwide (e.g. Boardman and Foster, 2008; Gutiérrez *et al*., 2009), gully erosion is often triggered and/or accelerated by inappropriate land use (LU). This trend is confirmed consistently for both sets of gullies. However, *cgs* (4.9% of LU3-5) are more prominent than *dgs* (2.6% of LU3-5) in cultivated areas and degraded grassland, whereas *dgs* (0.9% of LU1-2) are more prominent than *cgs* (0.6% of LU1-2) in natural vegetated and urban areas. The trend is not surprising since cultivated areas (LU3 and 4) and degraded grassland (LU5) represent areas where the soil is frequently disturbed and gully development (especially *cgs*) is favoured. Field observations indicate that a relatively large portion of the cultivated and grassland areas in the catchment is affected by gully erosion due to livestock disturbance, including overgrazing and trampling along cattle tracks.

Several studies identify the reduction in vegetation cover (VC) as the main driving factor of gully erosion (e.g. Tamene *et al.*, 2006; Descroix *et al.*, 2008). Figure 3 above indicates that gullies are mainly located in areas with poor VC. More specifically, *cgs* (2.4% of VC1) exceed *dgs* (1.4% of VC1) in areas with poor VC, whereas *dgs* (1.1% of VC2-5) exceed *cgs* (0.7% of VC2-5) in areas with moderate to good VC. Therefore, Figure 3 above illustrates that more vegetation is present in *dgs* than *cgs*. A probable reason is related to VC calculations being carried out in a grid-based system that depends on grid-cell resolution (Zhang *et al.*, 2002). For example, the Landsat TM image used to calculate the TSAVI and subsequent VC grid have a coarse resolution of 30 m^2 and therefore, small gullies with narrow patches of bare soil are incorrectly imbedded in vegetated areas (Taruvinga, 2008). Since *dgs* are frequently less than 30 m^2 in size, the proportion VC inside gullies at field scale could be overestimated, while the proportion bare soil could be underestimated.

Given that all zonal calculations in the study are based on a grid system, one of the main limitations of the study is that all outcomes will be subject to a certain degree of error. However, the variability between *cgs* and *dgs* caused by various grid-cell resolutions of the gully factor layers is outside the scope of current research and remains to be tested. It appears that the variability between scales is mainly affected by the high spatial heterogeneity of the study area itself. Another limitation worth mentioning here is that the study does not investigate land use history and vegetation conditions prior to gully development (since temporal scales are beyond the scope of this research). Therefore, uncertainties remain to what extent poor vegetation cover contributed to initial gully development in relation to other important contributing factors such as the intrinsic susceptibility of the soil. In effect, gully erosion processes itself can reduce the vegetation cover due to the removal of topsoil, as well as by soil tunneling/collapse. Nevertheless, similar to observations in a number of regions of South Africa (Laker, 2004; Le Roux *et al.*, 2008b), it is postulated that a combination of overgrazing and susceptible mudstones proves to be key factors that consistently determine the development of *cgs* and *dgs* in the catchment.

Conclusions and recommendations

Factors leading to the development of gullies in the catchment are consistent with other studies. However, previous research has not yet explicitly quantified differences in factor dominance between large continuous gullies (*cgs*) and relatively small discontinuous gullies (*dgs*). This factorial analysis contributes to perspectives on gully development by quantifying the differences or extent in factor dominance between *cgs* and *dgs*. The study indicates the complexity of a series of collective factors that are not identical between *cgs* and *dgs*. Factors leading to the development of *cgs* are gentle slopes in zones of saturation along drainage paths with a large contributing area, erodible duplex soils derived from mudstones, and poor vegetation cover due to overgrazing. When integrated with drainage networks, gullies expand from valley floors and footslopes onto concave midslopes where the soils are deep. Compared to continuous gully conditions, more *dgs* occur on rolling slopes where the surface becomes less frequently saturated with a smaller contributing area and where soils are more stable and shallow. These conditions prevent *dgs* from expanding extensively or from becoming continuous. However, they might still be active, as reported by Ndomba *et al.* (2009) for *dgs* in a catchment northeast of Tanzania. Further refinement will be possible given additional research, including investigation of the effect of land use history and vegetation conditions prior to gully development (e.g. Kakembo *et al.*, 2009), distinction between active and passive gullies using a combination of different optical and multi-temporal data (Ndomba *et al.*, 2009), and modelling gully erosion rates for representative test gullies and then averaging the results over the areas of active gully erosion (Flügel *et al.*, 2003).

Separation of gullies into these two groups is consistent with the findings of Descroix *et al.* (2008). The main difference to previous multi-scale studies such as Descroix *et al.* (2008) and Sonneveld *et al.* (2005) is specific quantification of the differences or extent in gully factor dominance between *cgs* and *dgs*. Some of the most prevalent differences between the two groups are apparent for the terrain unit and soil factors (land types and soil erodibility). A marked distinction can be made between large *cgs* favoured on footslopes with highly erodible soils (duplex and dispersive) and small *dgs* prevalent on a variety of terrain units with less erodible soils (that weather more slowly with minimal development). A combination of overgrazing and susceptible mudstones proves to be key factors that consistently determine the development of *cgs* and *dgs*.

Understanding the significance of gully controlling factors from field to catchment scale enables site– and scale–specific management intervention. For example, due to limited financial resources it will not be feasible to rehabilitate *cgs* with large and expensive structures at the catchment scale. However, it is imperative to minimize their current expansion from footslopes onto concave midslopes with site-specific construction of structures and protecting the vegetation from overgrazing (especially upslope along drainage paths situated on duplex soils). In addition to rehabilitating existing gullies, the identification of currently vegetated or gully-free areas susceptible to continuous and/or discontinuous gully development can also be achieved (not shown here - but estimated at approximately 560 and 6 700 ha, respectively). Appropriate strategies then need to be designed for susceptible areas in order to protect the current vegetation cover.

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Case Study III: Connectivity aspects in sediment migration modelling using SWAT

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Abstract

Sediment migration modelling at the catchment scale is complicated by various connectivity aspects between sources and sinks, including the extent that sediment generated on hillslopes is connected to a channel and linkage within a channel network. The Soil and Water Assessment Tool (SWAT) is applied within the context of connectivity in a catchment (Mkabela near Wartburg, South Africa) with identified source (cabbage plot) and sink (farm dams and wetlands) zones. The study illustrates SWAT can be applied in scenario analysis to assess connectivity aspects in sediment migration modelling. Scenario analyses establish the extent that sediment outputs from the cabbage plot create input for downstream sub-catchments, as well as the impact of farm dams and wetlands on sediment yield at the catchment scale. SWAT effectively identifies the cabbage plot as an important source of sediment at sub-catchment scale, but the sediment is not spatially identified within the sub-catchment where it is located and all the sediment is modelled to reach the channel, whether connected or not. Despite this, no significant changes are simulated by SWAT at the catchment outlet since increased discharge and sediment load from the cabbage plot is counterbalanced by sinks at the catchment scale. The model appears to be efficient in representing farm dams as a series of storages where connectivity is reduced at the catchment scale, but sediment deposited in farm dams mainly originates from surrounding sugarcane fields, not the cabbage plot. SWAT could not correctly identify wetlands as sink zones for cabbage sediment since, in contrary to farm dams, wetlands in SWAT are simulated off the main channel and water or sediment flowing into the wetlands must originate from the sub-catchment in which they are located. The suitability of SWAT for use in connectivity studies is discussed in the context of these findings.

Key words: Sediment connectivity, source-sink zones, SWAT model, catchment scale, South Africa.

Introduction

Soil erosion is a major soil degradation problem, confronting land and water resource management in many parts of the world (e.g. Prosser *et al.*, 2001; Lesschen *et al.*, 2009; Tibebe and Bewket, 2010). Besides the loss of fertile topsoil and reduction of soil productivity, soil erosion involves off-site impacts related to increased mobilization of sediment and delivery to rivers (Bracken and Croke, 2007). Water scarce countries such as South Africa are increasingly threatened by pollution and sedimentation of water bodies due to suspended sediment concentrations in streams which affects water use and ecosystem health (e.g. Flügel *et al*., 2003). It is imperative to devise the means through which these problems can be controlled but prevention and remediation relies largely on the understanding of factors controlling the sediment dynamics in a catchment, including sediment generation, transport and deposition (Lane *et al*., 1997; Parsons, 2012). The term connectivity is used to describe the extent to which sediment generated on hillslopes is connected to a channel by overland and subsurface flow, as well as the linkage of streamflow and sediment within a channel network (Hooke *et al*., 2003; Lesschen *et al*., 2009; Medeiros *et al*., 2010; Kakembo *et al*., 2012). Connectivity aspects from hillslopes to channels, as well as channel connectivity downstream needs to be considered. Good vegetation cover usually reduces connectivity from hillslopes to channels (Kakembo *et al*., 2012), whereas different sinks reduce connectivity within channels ranging from partial retention in small wetlands (Hatterman *et al*., 2006) to full blocking in large reservoirs (Medeiros *et al*., 2010). At the catchment-scale, connectivity aspects are driven by complex physical processes that involve interaction of a large number of spatial and temporal factors that cannot be monitored directly (Bracken and Croke, 2007).

Spatial and temporal variability poses a severe limitation, not only for local-scale measures, but also for procedures with a lumped nature, such as sediment rating curves and sediment delivery ratios that do not take connectivity aspects into account (Lenhart *et al*., 2005; Refsgaard and Hansen, 2010; Parsons, 2012). Assessments are usually carried out by means of a spatially-distributed sediment modelling approach (Collins and Walling, 2004), that accounts for connectivity aspects by integrating 2D-routing of sediment fluxes (Lenhart *et al*., 2005). Semi-distributed or semi-lumped models are often preferred above fully-distributed or physically-based models, since the application of the latter in large catchments lead to additional errors and uncertainty resulting from more parameters and input data requirements (Medeiros *et al*., 2010). The foundational strength of semi-distributed models is that they partition the catchment of interest into homogeneous morphological units thus, allowing to certain extents, the spatial variation of topography and land use to be accounted for (Lenhart *et al*., 2005; Gassman *et al*., 2007). Sediment migration at a catchment scale is often assessed by means of semi-distributed models such as the Soil and Water Assessment Tool (SWAT) (Arnold *et al*., 1998). Semi-distributed models such as SWAT, however, employ certain compromises or assumptions that disregard connectivity aspects (Lenhart *et al*., 2005).

In this context, this study aims to utilize SWAT to assess sediment migration and associated connectivity aspects at the catchment scale, including the influence of identified source and sink zones. The first objective is to model sediment migration with the SWAT model in a catchment (Mkabela near Wartburg, South Africa) with identified source and sink zones. Lorentz *et al*. (2011), by means of sediment fingerprinting, identified a cabbage plot in one of the upper sub-catchments as an important source of sediment, whereas farm dams and wetlands downstream function as sinks (details provided in the section below: Site description). The second objective is to investigate the suitability of SWAT for use in sediment migration modelling and connectivity studies by comparing model outputs with the sediment fingerprinting study of Lorentz *et al*. (2011). To our knowledge, previous studies have not applied and critiqued the SWAT model within context of connectivity. Our study provides insight into the applicability of SWAT in connectivity studies, specifically describing key strengths and weaknesses of the model when assessing sediment migration and catchment connectivity. Other implications of the study include supplementing the limited number of catchment-scale connectivity studies in general, as well as incorporation of small sediment sinks including farm dams and wetlands in catchment-scale modelling, an aspect neglected particularly in dryland agricultural regions, such as in South Africa. Although connectivity largely depends on rainfall duration and intensity to produce connected flow or transport of sediment (Bracken and Croke, 2007), SWAT is not designed as a field-scale event-based model. Therefore, the emphasis herein is on annual average results on sediment migration as represented by the SWAT model's spatial elements including sub-catchments and catchment. Our discussion focuses on a spatial scale beyond the variability of infiltration and we do not consider the influence of subsurface flow on connectivity due to the lack of appropriate data.

Materials and methods

Site description

The Mkabela catchment lies between 29º 21' 12'' and 29º 27' 16'' south and 30º 36' 20'' and 30º 41' 46'' east in the KwaZulu-Natal Province of South Africa, northeast of the town Pietermaritzburg (see Figure 1). Elevation ranges from 880 m at the catchment outlet in the southwest to 1 057 m upstream in the northeast of the catchment. The catchment area of 4 154 ha is drained by a tributary of the Mgeni River with a flow length of approximately 12.6 km from its source to the catchment outlet. Connectivity is influenced by a series of 9 farm dams and 5 wetlands along the axial valley, ranging between 0.6-10 and 5.4-22 ha, respectively (see Table 1 and Figure 2 in the Model input section). Landforms are complex, ranging from gently undulating footslopes and valley floors to very steep midslopes exceeding 20%. The climate is sub-humid with a mean annual rainfall of 825 mm of which around 80% is recorded in the summer season extending from October to April. The mean annual potential evaporation is 680 mm, as estimated by the Priestley and Taylor (1972) method in SWAT. July is the coolest month whereas February is the warmest month with mean minimum and maximum temperatures ranging from 6 to 21ºC and 17 to 28ºC, respectively.

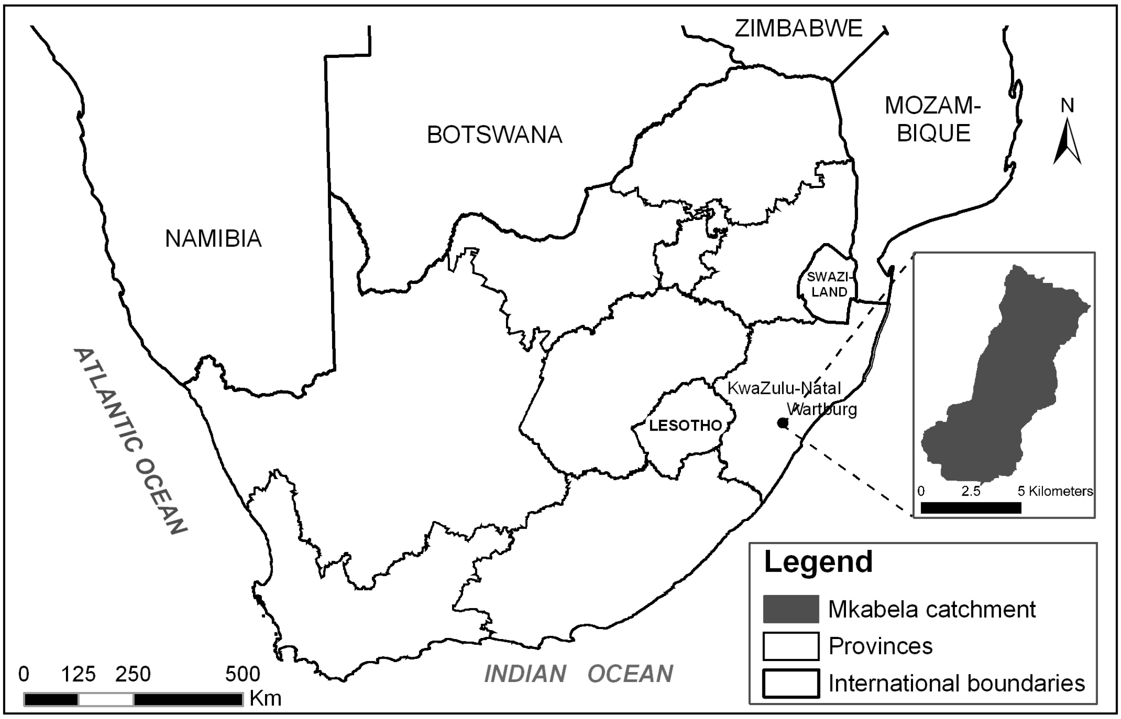

Figure 1: **Location map of Mkabela catchment in the KwaZulu-Natal Province, South Africa.**

The geology consists of sandstone of the Natal Group of the Cambrian Age and a relatively small pocket of Ecca sedimentary rocks in the north (Council for Geoscience, 2007). Soils vary from poorly drained clays predominately in the northern part of the catchment and areas with low relief (e.g. Westleigh form) to well drained sandy soils mainly in the southern part of the catchment in areas with high relief and steep slopes (e.g. Hutton form) (Land Type Survey Staff, 1972-2008). The major soil types occur in the central part of the catchment, including shallow sandy soils on steep and convex hillslopes with little water holding capacity (Cartef form occupying approximately 36% of the catchment) and deeper sandy soils on midslopes with soft or hard plinthic sub-horizons that is permeable to water (Glencoe and Avalon forms occupying approximately 20% of the catchment). The catchment falls within the Savanna Biome (Mucina and Rutherford, 2006) but natural vegetation in the catchment has been replaced or modified by agricultural activities several decades ago. Most of the catchment is under sugarcane cultivation (3 100 ha or 75% of the catchment) with minority land uses including forestry (13%), pasture (8%) and a cabbage plot (3%). Lorentz *et al.*

(2011), by means of sediment fingerprinting, identified the cabbage plot in sub-catchment 1 as an important source of sediment, whereas farm dams and wetlands downstream function as sinks. The wetland in sub-catchment 11 is the major sink for the cabbage sediments (Figure 2).

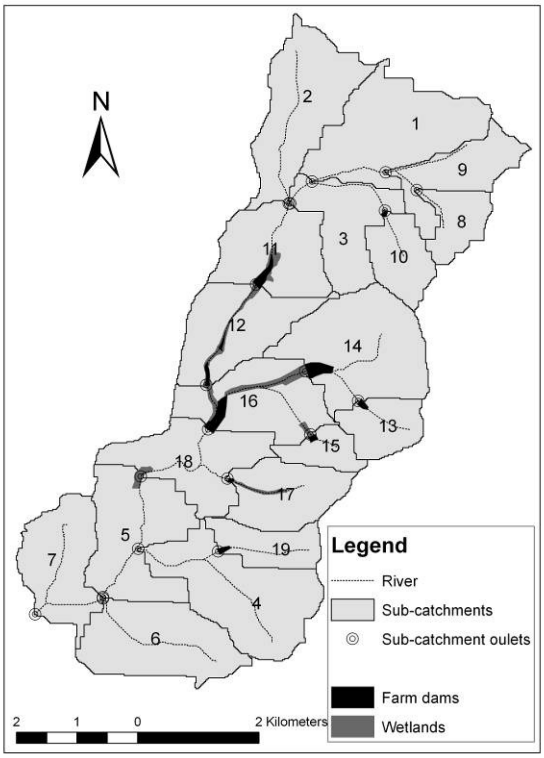

Figure 2: **Sub-catchment boundaries, outlets, location of river channel, farm dams and wetlands.**

Model selection and description

The Soil and Water Assessment Tool (SWAT) was selected mainly because it is a spatially semi-distributed model that has gained international acceptance and has been applied to support various large catchment (10–10 000 km^2) modelling studies across the world with minimal or no calibration effort (e.g. Mishra *et al*., 2007; Wang *et al*., 2009; Srinivasan *et al*., 2010). The foundational strength of SWAT is that it considers most connectivity aspects into one simulation package, including factors controlling upland sediment generation, channel transport and deposition into sinks (Gassman *et al*., 2007). Furthermore, SWAT is routinely coupled with geographical information systems which, according to Chen and Mackay (2004), offer unprecedented flexibility in the representation and organization of spatial data.

SWAT is a catchment-scale, continuous time model operating on a daily time-step developed by the US Department of Agriculture (USDA) Agricultural Research Service to simulate water, sediment and chemical fluxes in large catchments with varying climatic conditions, soil properties, stream channel characteristics, land use and management practices (Arnold *et al*., 1998; Srinivasan *et al*., 1998). First, a catchment is divided into multiple sub-catchments, which can be further divided into hydrological response units (HRUs) consisting of homogeneous soil and land use characteristics (Gassman *et al*., 2007). The hydrologic component is based on the water balance equation in the soil profile integrating several processes, including surface runoff volume using the Green and Ampt (1911) infiltration method or the USDA SCS (1972) curve number method. Here, the SCS curve number method was chosen which is empirically based and relates runoff potential to land use and soil characteristics. Peak runoff rate is estimated with a modification of the rational method, where runoff rate is a function of daily surface runoff volume and a proportion of rainfall occurring until all of the catchment is contributing to flow at the outlet, known as the time of concentration (Neitsch *et al*., 2005). The time of concentration is estimated using Manning's Formula considering both overland and channel flow. Sediment yield caused by rainfall and runoff is computed with the Modified Universal Soil Loss Equation (MUSLE) (Williams, 1975), using surface runoff and peak flow rate together with the widely used USLE (Wischmeier and Smith, 1978) factors including soil erodibility, slope length and steepness, crop cover management and erosion control practice. Certain nutrients and pesticides are also simulated by SWAT, but are outside the scope of this research and are not described here.

Once the loadings of water and sediment have been determined, they are summed to the sub-catchment level and routed through the stream network of the catchment including ponds, wetlands, depressional areas, and/or reservoirs (Neitsch *et al*., 2005). SWAT incorporates a simple mass balance model to simulate the transport of sediment into and out of water bodies, where settling is calculated as a function of concentration and transportation out of a farm dam is a function of the final concentration (Neitsch *et al*., 2005). Flow is routed through the channel using either the variable-rate storage method (Williams, 1969) or the Muskingum method (Overton, 1966), which are both variations of the kinematic wave model. Here the default variable storage method was chosen. Sediment is routed by means of a simplified stream power theory where the maximum amount of sediment that can be transported, deposited or re-entrained from a channel segment is a function of the peak channel velocity (Arnold *et al*., 1995). The equations mentioned above and additional theoretical documentation for SWAT is given by Neitsch *et al*. (2005). AVSWAT-X which is a graphical user interface for SWAT and ArcView® software extension (Di Luzio *et al*., 2004) was used for this study. A description of the input data requirements follows.

Model input

The AVSWAT-X interface requires several spatial datasets including topography, drainage network, land cover, soil, climate and land management. First, topographic and drainage network data were prepared from a digital elevation model (DEM) with a grid cell resolution of 20 m (GISCOE, 2001). Automated routines in AVSWAT-X calculated the slope and divided the catchment into sub-catchments from the DEM. Appropriate contributing source areas and sub-catchment sizes had to be established by the user as percentage area of the entire catchment, i.e. 30%. Several studies reviewed by Gassman *et al.* (2007) suggest setting sub-catchment areas at much smaller percentages (<5% of the catchment) to ensure accuracy of estimates, but such values are not feasible for larger catchments as simulated in this study. The number of sub-catchment links or outlets was manually adjusted, representing all the relevant tributaries of the main river into 19 sub-catchments that are comparative in size, as well as to ensure that flow monitoring points spatially overlay with sub-catchment outlet points for calibration of model simulations with field measurements. Thus, each of the 19 sub-catchments consists of a channel with unique geometric properties not shown here including slope gradient, length and width. Manning's roughness coefficient was assigned to each segment in order to represent conditions observed in the field. Channel erosion parameters were set to default representing non-erosive channels due to the lack of data but also to eliminate channel erosion in simulations. According to observations, most sediment is generated from agricultural fields (Lorentz *et al.*, 2011). Gullies are absent in the Mkabela catchment so that the simulated sediment yields could be interpreted according to the empirical soil loss equation MUSLE used, which does not account for gully erosion.

In addition, 9 outlets were incorporated to represent outlets at the exit from 9 farm dams. AVSWAT-X also allows relatively small impoundments such as wetlands to receive loadings from a fraction of the sub-catchment area where it is located. Figure 2 illustrates the geographical distribution and extent of the farm dams and wetlands digitized from SPOT 5 panchromatic sharpened images at 2.5 m resolution acquired in 2006, whereas Table 1 contains parameter information obtained from Le Roux *et al.* (2009). The discretisation resulted in the definition of 19 sub-catchments that are joined by outlets and tributary channels branching off the main channel, including 9 farm dams and 5 wetlands along the axial valley.

Table 1: Parameter information used to model each of the farm dams and wetlands in Mkabela catchment (adapted from Le Roux *et al*., 2009).

Sub-catchment	Dam area (ha)	Dam volume (m^3)	Wetland area (ha)	Wetland volume (m^3)
5	-	-	5.44	108 800
10	0.7	11 800	-	-
11	5.9	229 600	4.73	141 900
12	4.5	87 000	9.17	183 400
13	1.7	31 800	-	-
14	8.4	330 400	-	-
15	1.5	26 600	-	-
16	10.3	405 600	22.44	673 200
17	1.2	20 400	4.88	97 600
19	2.5	47 800	-	-

Next, a land cover map was digitized from SPOT 5 imagery acquired in 2006, followed by ground truthing (see Figure 3a). The land cover map was linked to a database in AVSWAT-X consisting of several plant growth parameters. The plant growth component of SWAT is a simplified version of the EPIC plant growth model (Sharpley and Williams, 1990), where phenological plant development is based on daily accumulated heat units developed by Monteith (1977) and biomass is inhibited by temperature, water or nutrient stress. SWAT also requires information on soil properties that govern the movement of water through the soil profile. An unpublished pedological soil map at a scale of 1:100 000 with textural profile descriptions for all major soils was used (Le Roux *et al*., 2006) (see Figure 3b).

In order to improve the display and representation of the variable soils in the catchment, the major soil units were delineated into smaller terrain units by means of the topographical algorithms of Evans (1979) and Schmidt *et al*. (2003). To account for soil variability with depth, up to three layers/horizons were incorporated into each soil component. Textural parameter values were assigned to each unit and layer according to the textural profile descriptions given by the soil map. Pedotransfer functions similar to Tol *et al*. (2010) were used to generate the required hydraulic parameters, including available water capacity and saturated hydraulic conductivity. The overlay of land cover and soil maps created 130 hydrological response units (HRUs). These are portions of a sub-catchment that possess unique land use and soil attributes. Similar to Bouraoui *et al*. (2005), the discretisation was done to keep the number of HRUs down to a reasonable number, while considering the diversity and sensitivity of land cover and soil combinations. The study aimed at integrating all land cover units that significantly affect the sediment yield of a catchment, whether large or small in spatial extent.

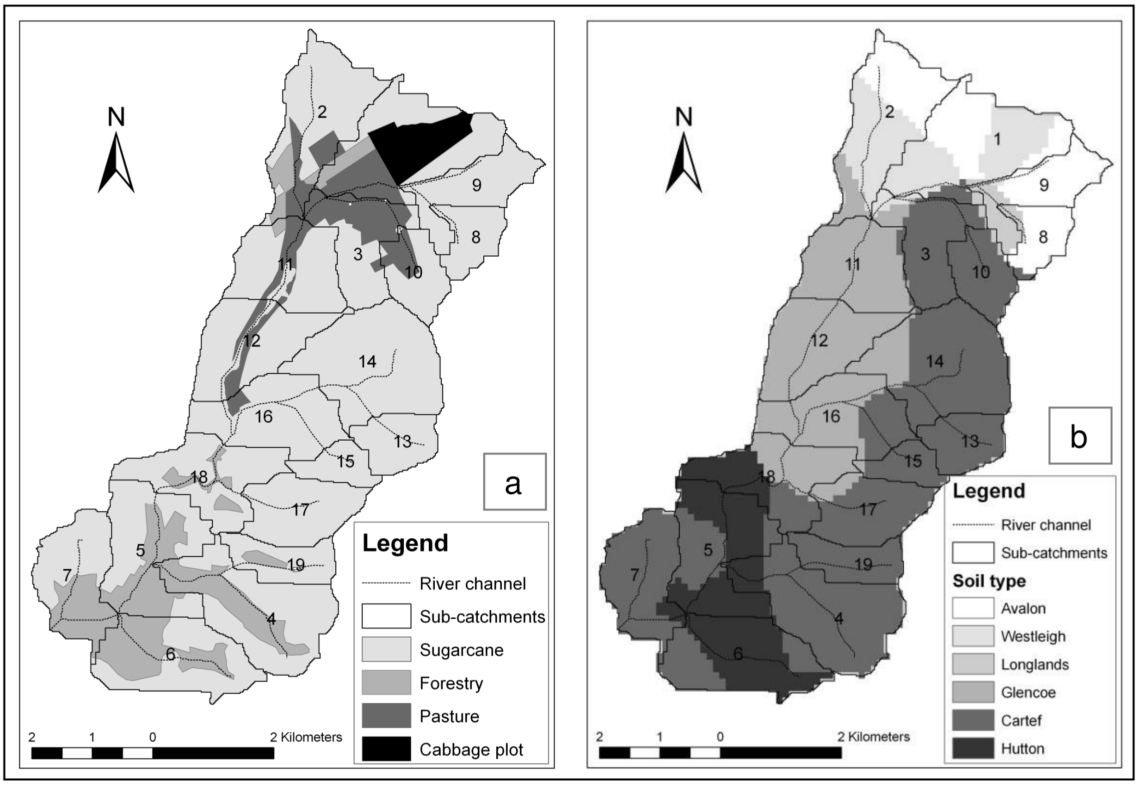

Figure 3: **(a) Land cover map and (b) soil map of Mkabela catchment (after Le Roux *et al.*, 2006).**

AVSWAT-X also requires spatial data for several climate parameters including precipitation, temperature, solar radiation, relative humidity and wind speed. These were calculated from daily values over a 30 year period (1 January 1977 to 30 June 2008) from 4 stations within 2 kilometres or less of the catchment boundary (Agrometeorology Staff, 1984-2008). Since not all the stations have full records of the required parameters, incomplete records were patched with the most complete and closest stations. Finally, ancillary information regarding management practices in the catchment was incorporated including tillage operations, nutrient applications, irrigation scheduling and harvesting operations. Due to the lack of data on crop rotation systems and timing of agricultural operations, phenological plant development is based on daily accumulated heat units. Detailed descriptions of the parameters are provided by Neitsch *et al.* (2005).

Model calibration and validation

Calibration and validation were restricted to measurements from an ISCO sampler and H-flume at the outlet of sub-catchment 8 (area of 96 ha) from August 2006 to March 2008, including sediment loads of 5 rainfall events between October 2007 to January 2008. Calibration of SWAT focused mainly on the hydrological part of the model on a monthly time-step adjusting the most sensitive model parameters similar to other studies (e.g. Bouraoui *et al.*, 2005; Tibebe and Bewket, 2010). The hydrological component was calibrated by modifying the curve number and base-flow coefficients, whereas the erosion component was calibrated by adjusting the MUSLE soil erodibility and support management factors. Model performance was improved by sequentially optimizing the widely used coefficient of efficiency (E) of Nash and Sutcliffe (1970), as well as the coefficient of determination (r^2). As a measure of goodness-of-fit between simulated and observed loads, a simple per cent deviation method of Martinec and Rango (1989) was used; given as:

$$Dv = [V - V' / V] \times 100$$

where, V is the measured runoff volume and V' is the simulated volume. Dv will be zero for a perfect fit and the smaller the value the more accurate are the simulated results. Subsequently, it was possible to hydrologically calibrate the model at the flume by sufficiently tracking the average monthly trends during the simulation period. Overall, SWAT over-predicted discharge by 6.2% as determined by Dv. The goodness of fit expressed by E was 70% and r^2 was 82%, indicating a close relationship between the observed and simulated discharge. Figure 4 illustrates the observed and simulated discharge and sediment loads of 5 rainfall events that occurred during October 2007 to January 2008. Although the limited number of observed events cannot be used to fully validate the model, there is good indication that a large part of the suspended sediment load can be explained by event discharge and that the model is able to track the loads of these events at least within the order of magnitude of observed values.

The observed data were inadequate to validate total discharge and sediment yield at the catchment outlet. Unfortunately, a major limitation to the use of continuous time models such as SWAT in developing countries is the lack of recorded flow and sediment data for calibration and validation (Van Zyl, 2007). Nevertheless, similar to the study of Bouraoui *et al.* (2005), calibrated values for specific HRUs in sub-catchment 8 were extended downstream to the larger catchment area with corresponding HRUs under sugarcane cultivation. More detail on these HRUs is outside the scope of the text. Although accurate predictions were not the goal of this study, the calibration, together with above-mentioned fingerprinting study of Lorentz *et al.* (2011), served to establish a realistic baseline for spatially extending the AVSWAT-X model downstream in order to investigate connectivity aspects of sediment migration modelling.

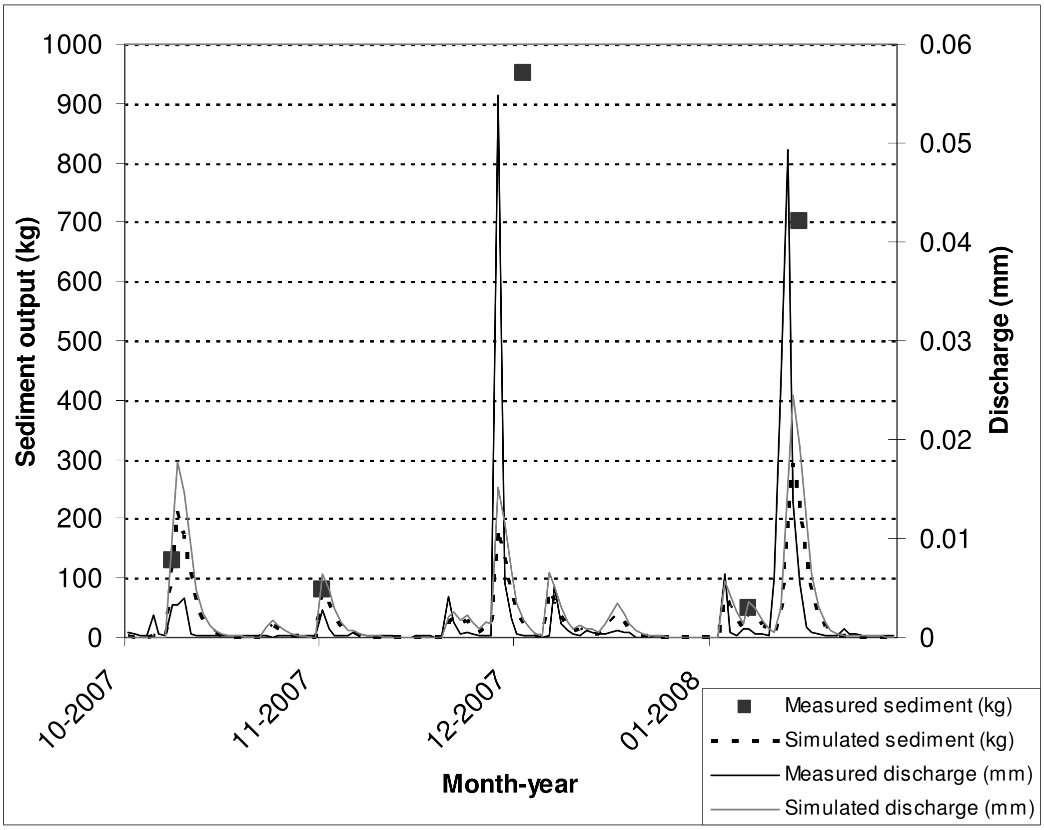

Figure 4: **Observed and simulated discharge and sediment loads of 5 rainfall events that occurred from October 2007 to January 2008.**

Connectivity aspects in sediment migration

Central to this study was the assessment of connectivity aspects in sediment migration at the catchment scale with the SWAT model. In order to create a catchment overview of sediment migration downstream and the associated connectivity aspects, the study performed four additional simulations with the AVSWAT-X model after simulating the observed catchment condition with all dams and wetlands in place. Two scenarios were performed to establish the extent that sediment outputs from the identified sediment source (cabbage plot) create input in addition to sugarcane for downstream sub-catchments, whereas another two scenarios were performed to establish the impact of existing sinks (9 farm dams and 5 wetlands) on connectivity downstream. In total, 5 simulations were conducted over a period of 2 years (1 July 2005 to 30 June 2008) preceded by a one-year model "warm-up" initialization period. The four scenarios are summarized as follows:

- 1a Replacing the current cabbage plot with sugarcane;
- 1b Replacing existing pasture and sugarcane fields in sub-catchment 1 with cabbage, subsequently expanding the current cabbage plot by approximately 300% (from 114 to 351 ha) and connecting it with the main channel;
- 2a Simulating current conditions without farm dams;
- 2b Simulating current conditions without wetlands.

The results for each scenario were scrutinized for changes in the simulated sediment outputs from the upper to lower sub-catchment outlets, including dams and wetlands along the main river. This was mainly achieved by investigating the annual changes in simulated discharge and sediment loads as represented by the model's spatial elements, namely sub-catchments and catchment.

Results and discussion

The migration of sediment and associated connectivity aspects in the Mkabela catchment are described here. Results of the current conditions as simulated by SWAT are followed by scenarios evaluating the influence of identified source and sink zones on connectivity at the catchment scale. Results from the scenario analysis reveal the progress of discharge and sediment migration downstream and associated connectivity aspects at the catchment scale. Finally, a discussion on the suitability of SWAT for use in connectivity studies is provided.

Sediment dynamics in the Mkabela catchment

Figure 5 illustrates the sediment yield in t/ha for each sub-catchment that is transported into the channel during the observation period (1 July 2006 to 30 June 2008). Results substantiate the findings of Lorentz *et al*. (2011) that sub-catchment 1 containing the cabbage plot is a significant sediment source. Although sub-catchment 1 is characterized by flat slopes between 0 and 2%, sediment yield (1.7 t/ha) is several orders of magnitude larger than yields (0.001 t/ha) in sub-catchments downstream (e.g. 4, 5, 6 and 7) with steep slopes up to 30%. The main reason for this discrepancy is related to vegetation cover. Latter sub-catchments contain sugarcane and forestry plantations with good seasonal groundcover, whereas sub-catchment 1 contains a cabbage plot with relatively poor groundcover. Furthermore, soil under the cabbage plot consists of poorly drained clays that are more prone to runoff and erosion than the well-drained sandy soils of sub-catchments 4, 5, 6, and 7.

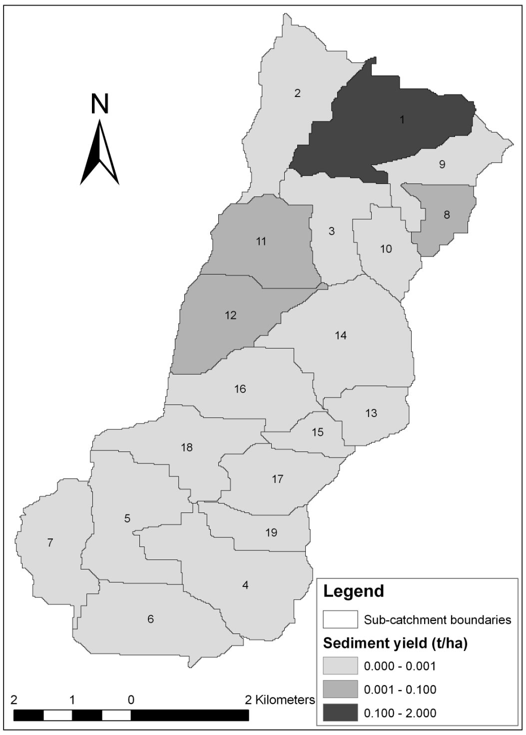

Figure 5: **Sediment yield per sub-catchment (in t/ha) that is transported into the channel during the observation period (1 July 2006 to 30 June 2008).**

Figure 6a illustrates the monthly average streamflow in m^3/s for 9 sub-catchments connected with the main channel and Figure 6b shows the total sediment in metric t per month transported out of their outlets. Streamflow ranges between 0.003 m^3/s at sub-catchment 1 in September 2007 to 0.701 m^3/s at the main catchment outlet in January 2008, whereas sediment loss ranges between 0.059 t at sub-catchment 9 in September 2007 to 19.46 t at the main catchment outlet in January 2008. Model outputs substantiate several logical criteria regarding sediment dynamics that are consistent with studies in other parts of the world. First, Figure 6a and 6b follow the same pattern which indicates that sediment output is controlled by the water flux. Second, results clearly illustrate a summer dominant erosion pattern which is mainly caused by intensive summer rainfall totalling 620 mm between October and April. According to simulations, nearly 70% of the average annual streamflow and over 85% of the annual sediment output (approximately 70 metric t per annum) are concentrated in the rainy season. Third, low outputs occur mainly in the upper sub-catchments and increase downstream due to the cumulative contribution of runoff and sediment routed downstream from sub-catchment outlet 8 to 7. The migration of sediment downstream explains why certain sub-catchments with relatively low sediment yields (see Figure 5) have high sediment outputs at their outlets (see Figure 6b) and vice versa. The following section discusses the impacts of identified source and sink zones on connectivity as simulated by SWAT.

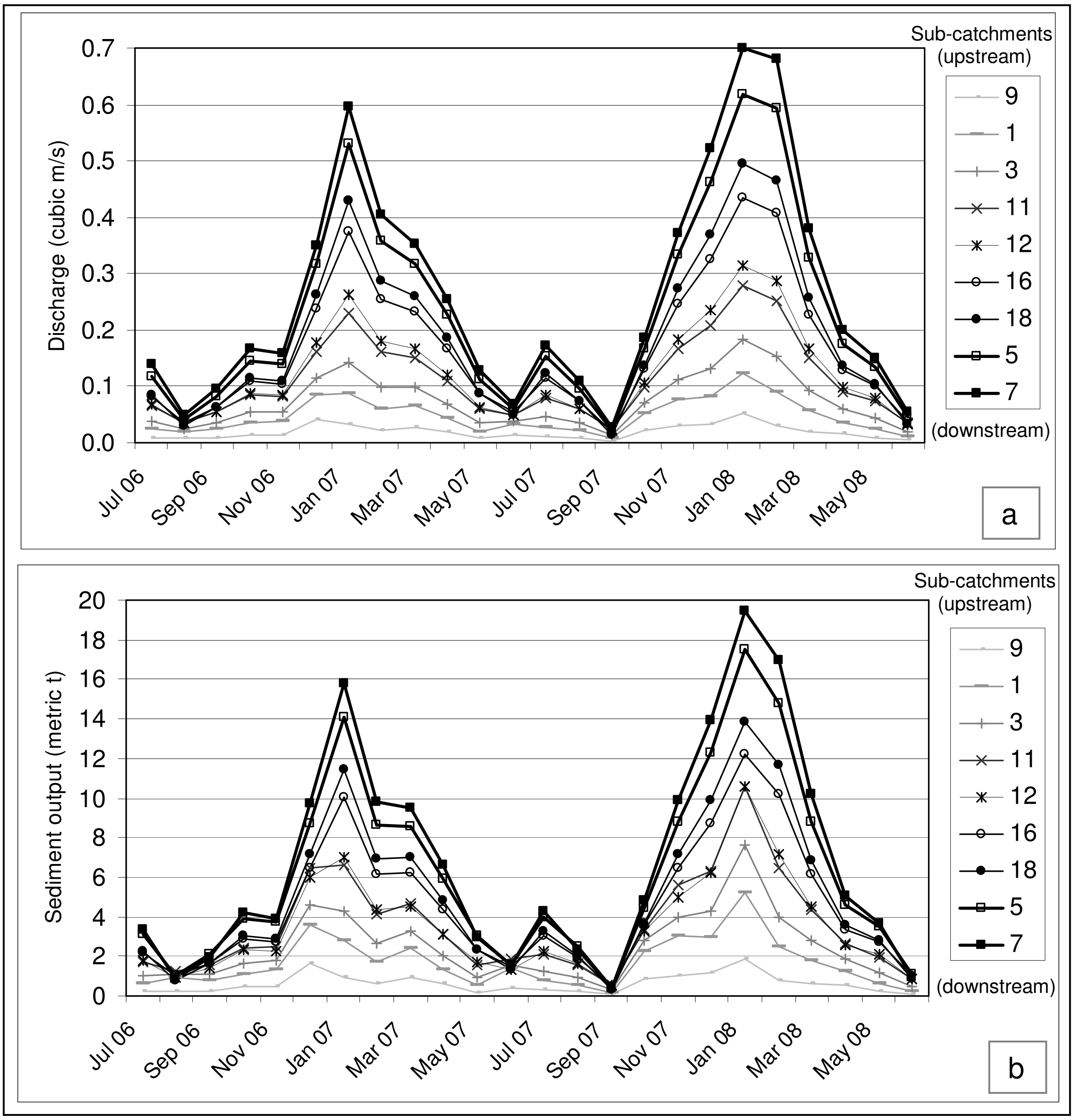

Figure 6: **(a) Monthly average streamflow (m^3/s) for 9 sub-catchments connected with the main channel; (b) Total sediment (metric t per month) transported out of the 9 sub-catchments. Sub-catchment numbers are assigned arbitrarily.**

Scenarios assessing the influence of identified source and sink zones

Scenario impacts on discharge and sediment output for 9 sub-catchments along the main channel are illustrated in Figure 7. Scenarios 1a and 1b illustrate the extent that sediment outputs from the identified sediment source (cabbage plot) create input in addition to sugarcane for downstream sub-catchments, whereas scenarios 2a and 2b establish the impact of existing sinks (9 farm dams and 5 wetlands) on connectivity downstream. Impacts are expressed as the percentage difference between current conditions and four scenarios assessing the influence of the identified source and sink zones.

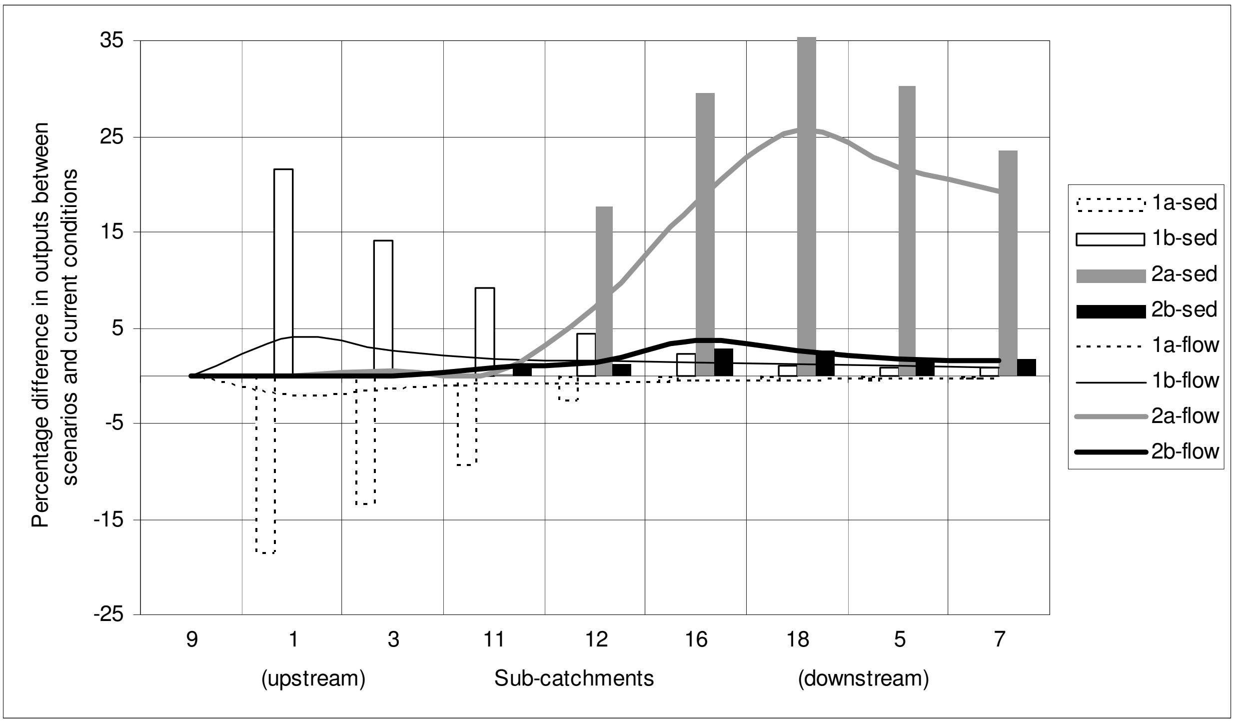

Figure 7: **Scenario impacts on discharge and sediment output for 9 sub-catchments long the main channel, expressed as the percentage difference between current conditions and four scenarios (1a - replacing cabbage plot with sugarcane; 1b – expanding cabbage plot by approximately 300%; 2a - current conditions without farm dams; 2b - current conditions without wetlands).**

Scenario 1a: Replacement of cabbage plot with sugarcane

Replacement of the cabbage plot with sugarcane illustrates the extent that sediment outputs from the cabbage plot create input in addition to sugarcane for downstream sub-catchments. Figure 7 illustrates that replacement of the cabbage plot with sugarcane decreases average annual discharge and sediment output the most at the outlet of sub-catchment 1 containing the cabbage plot (-2.2% and -18.4% respectively) and reduces downstream (to -0.4% and -0.2% respectively at the main catchment outlet). Results indicate that the cabbage plot increases discharge and sediment output the most at sub-catchment 1 in which it is located and impact on discharge and sediment output diminishes downstream to nearly zero percent past sub-catchment 11. During the simulation period, sediment from the cabbage plot is deposited downstream mainly in the channel along sub-catchments 1, 3 and 11 (approximately 287.6 t/yr). The main reason for deposition in the channel is its relatively rough surface conditions with a Manning's roughness coefficient of 0.1 (a typical value assigned to grass covered channel beds with good riparian vegetation). Another reason is the limited number of large rainfall events and associated peak channel velocities needed to transport and re-entrain sediment during the simulation period. Sediment from the cabbage plot that is not deposited in the channel is deposited in the farm dam of sub-catchment 11 (approximately 7.4 t/yr). The following scenario illustrates the effect of a larger cabbage plot on annual changes in simulated discharge and sediment loads.

Scenario 1b: Cabbage plot expanded

In Scenario 1b, the cabbage plot is expanded approximately 300% so that it covers the whole of sub-catchment 1 at the expense of sugarcane and pasture. As expected, Figure 7 illustrates that the average annual discharge and sediment output increase the most at the outlet of sub-catchment 1 (4.0% and 21.6% respectively) and reduces downstream (to 0.8% and 0.9% respectively at the main catchment outlet). Similar to scenario 1a, the impact of the larger cabbage plot on discharge and sediment output diminishes downstream due to deposition along the channel of sub-catchments 1, 3 and 11 (approximately 760.0 t/yr), including the farm dam of sub-catchment 11 (approximately 13.0 t/yr). Compared to scenario 1a, however, outputs nearly triple and more sediment migrates beyond sub-catchment 11. This is reasonable given that there is a greater supply of sediment since cabbage has less groundcover than sugarcane and pasture, but also supposedly uninterrupted connectivity between the cabbage plot and channel. The expanded hydrological response unit covers the whole sub-catchment 1 which is locationally connected to simulated channel – although this is not accounted for in SWAT and is discussed below. The following section discusses the impact of identified sink zones on connectivity as simulated by SWAT.

Scenario 2a: Removing farm dams

Figure 7 illustrates that removal of farm dams increase discharge and sediment output the most at sub-catchment outlets downstream (11 downwards) where most of the farm dams normally occur. More specifically, average annual discharge and sediment output increase the most at the outlet of sub-catchment 18 (25.6% and 36.7% respectively). Although sub-catchment 18 does not contain a farm dam within its boundaries, seven farm dams are located near and upstream of it, subsequently retaining its loadings as illustrated in Figure 7. In relation to the amount of discharge and sediment reaching the catchment outlet, 19.2% and 23.5% is retained respectively. Since nearly all sediment from the cabbage plot is deposited in sub-catchments 1, 3 and 11 before reaching farm dams downstream, sediment deposited in farm dams mainly originates from surrounding sugarcane fields. According to the simulation, average sediment deposition per farm dam equals 6.3 t/yr. Although studies elsewhere report more drastic declines in sediment retention in dams (e.g. 64% by Mishra *et al*., 2007), our results seem reasonable given the fact that the farm dams are relatively small with an average storage capacity of 136 333 m^3 and regularly spill, thus frequently releasing suspended sediment (Medeiros *et al*., 2010). Importantly, results were able to represent farm dams as a series of storages where flow is reduced, sediment deposited and thus connectivity is reduced.

Scenario 2b: Removing wetlands

The impact of wetlands is investigated by simulating current conditions without wetlands. Figure 7 illustrates that removal of wetlands increase discharge and sediment output at sub-catchment outlets where most of the wetlands would occur. More specifically, average annual discharge and sediment output increase the most at the outlet of sub-catchment 16 (3.6% and 2.8% respectively) containing the largest wetland. At the catchment outlet, discharge is reduced by 1.6% and sediment output by 1.7%. Average sediment deposition per wetland equals 0.012 t/yr. Compared to current conditions without farm dams in scenario 2a, wetlands influence discharge and sediment output insignificantly, subsequently influence connectivity less than farm dams. These results remain questionable since the wetlands have a larger total area and storage capacity (47 ha and 1 204 900 m^3) compared to farm dams (36 ha and 1 191 000 m^3), and since the fingerprinting study of Lorentz *et al*. (2011) established that the wetland in sub-catchment 11 is a major sink for the cabbage sediments. The following section provides a brief discussion of how model compromises or assumptions affect outputs in the context of connectivity between sources and sinks, as represented by the model's spatial elements including sub-catchments and catchment.

Suitability of SWAT in connectivity studies

In terms of source zones at the sub-catchment scale, SWAT simulations substantiate the findings of Lorentz *et al*. (2011) that the cabbage plot is an important sediment source in the Mkabela catchment (see Figure 5). Results illustrate, however, that sediment generated on the relatively small cabbage plot is not spatially identified within the sub-catchment it is located. The whole of sub-catchment 1 is highlighted as a source in Figure 5. The non-spatial aspect of hydrological response units (HRUs) ignores flow and sediment routing within a sub-catchment (Gassman *et al*., 2007). SWAT does not consider the processes of deposition during transport from HRUs to channel since all the eroded sediment calculated for the separate HRUs reaches the channel (Lenhart *et al*., 2005). Comparison of the SWAT output tables of scenario 1a and current conditions not shown here indicates that all the sediment generated from the cabbage plot reaches the channel of sub-catchment 1. Likewise, the increase in percent discharge and sediment loads shown in Figure 7 for scenario 1b can be explained exclusively by the increase in the spatial extent of the cabbage plot at the expense of sugarcane and pasture, not due to its uninterrupted connectivity with the channel. In reality the potential for different HRUs of a sub-catchment to contribute to sediment yield is controlled by a complex interplay of connectivity aspects including locational and filter resistance during transport from HRUs to channel (Collins and Walling, 2004). The HRU approach in SWAT disregards processes of deposition in the pasture HRUs between the cabbage plot and channel. Although filter strips and field borders can be simulated at the HRU level based on empirical functions, assessments of targeted filter strip placements or riparian buffer zones is not possible due to the lack of HRU spatial definition in SWAT (Gassman *et al*., 2007). Chen and Mackay (2004) suggest the use of smaller sub-catchments instead of HRUs, but this approach is only applicable in small catchments and the impacts on SWAT output as a function of variation in HRU and/or sub-catchments is outside the scope of the study. Chen and Mackay (2004) provide further detail on the extent predictions in general are altered by using HRUs, as well as the mechanisms by which sediment is moved from sub-catchments to channels.

In terms of sink zones at the catchment scale, SWAT seems to be particularly efficient in representing the farm dams as a series of storages where flow is reduced, sediment deposited and thus connectivity is reduced (see Figure 7). However, the impact of farm dams on connectivity needs further investigation in the Mkabela catchment since no measurements have been made on sediment input and output from farm dams. Results are based on the assumptions in SWAT that water bodies are completely mixed and sediment is instantaneously distributed throughout the volume at entering. SWAT could not effectively identify wetlands as sink zones and simulations do not correlate with the findings of Lorentz *et al*. (2011) that cabbage sediment is primarily deposited in the wetland of sub-catchment 11. As mentioned above, during the simulation period of 2 years large portions of cabbage sediment is deposited in the channel along sub-catchments 1, 3 and 11 – here the channel in effect acts as a wetland due to its relatively rough surface conditions with a Manning's roughness coefficient of 0.1 (a typical value assigned to grass covered channel beds with good riparian vegetation). However, not even smoother channel conditions or a longer simulation period will ensure that sediment from the cabbage plot is transported to the wetland in sub-catchment 11. Wetlands simulated by SWAT only retain the water and sediment originating from the sub-catchment within which they are located. Wetlands in SWAT cannot receive and retain loadings from the sub-catchment upstream containing the cabbage plot. In contrary to the way farm dams are simulated, wetlands are simulated off the main channel and water or sediment flowing into them must originate from the sub-catchment in which they are located (Neitsch *et al*., 2005). This largely explains why the percentage difference in discharge and sediment output of scenario 2b without wetlands are less than that of scenario 2a without farm dams (see Figure 7). Although changing the model structure was not an objective of the study, modifications of SWAT (Wang *et al*., 2009) or application of other SWAT-based models such as SWIM (Hatterman *et al*., 2006) where wetland processes are incorporated more realistically may greatly improve simulation of wetland dynamics.

Conclusions and recommendations

The Soil and Water Assessment Tool (SWAT) is applied within the context of connectivity in the Mkabela catchment in KwaZulu-Natal, South Africa, including the influence of identified source and sink zones. The study illustrates how the model can be applied in scenario analyses to assess connectivity aspects in sediment migration modelling. Scenario analyses establish the extent that sediment outputs from the identified sediment source (cabbage plot) create input for downstream sub-catchments, as well as the impact of major sinks (9 farm dams and 5 wetlands) on sediment yield downstream. Results are consistent with other studies where vegetation cover and soil type of source zones have major influences on sediment generation (e.g. Medeiros *et al.*, 2010), whereas structures such as farm dams serve as important sink zones where sediment is deposited (e.g. Mishra *et al.*, 2007). More specifically, the cabbage plot is an important source of sediment because of relatively poor seasonal groundcover and poorly drained clays prone to runoff and erosion. The removal and expansion of the cabbage plot in our scenario analyses significantly changes discharge and sediment yield upstream. However, similar to the studies of Walling (1999) and Prosser *et al.* (2001), no significant changes are simulated at the catchment outlet. The main reason is the channel serves as an important sink zone due to its relatively rough surface conditions with a Manning's roughness coefficient of 0.1 (a typical value assigned to grass covered channel beds with good riparian vegetation). Although the removal of farm dams significantly changes discharge and sediment loads at the catchment outlet, sediment deposited in farm dams mainly originates from surrounding sugarcane fields, not the cabbage plot. Subsequently, the effect of sediment sinks becomes dominant over sediment sources with increasing spatial scale as addressed by several other studies (Kirkby *et al.*, 1996; De Vente and Poesen, 2005; De Vente *et al.*, 2007; Lesschen *et al.*, 2009). In order for results to be useful for site– and scale–specific management intervention, it is important to consider how model compromises or assumptions affect outputs in context of connectivity between sources and sinks, as represented by the model's spatial elements including sub-catchments and catchment.

At the sub-catchment scale, SWAT effectively identifies the cabbage plot as an important source of sediment. However, cabbage plot sediment is not spatially identified within the sub-catchment it is located and all the sediment generated from the plot reaches the channel, whether connected to the channel or not. A major weakness of SWAT is that it does not consider the processes of deposition during transport from hillslopes/HRUs to channel (Lenhart *et al*., 2005). Chen and Mackay (2004) suggest the use of smaller sub-catchments instead of HRUs, but this approach is not applicable in large catchments such as simulated here. In large catchments, discretisation should be done to keep the number of sub-catchments and HRUs down to a reasonable number, while considering the diversity and sensitivity of land cover and soil combinations. It is recommended that future SWAT-based research determine how catchment partitioning affects model outputs in the context of connectivity. Such research will require assessments at relatively fine spatial and temporal scales, including factors influencing connectivity at hillslope/HRU scale such as processes of overland/subsurface flow and site-specific process zones in channels, and relationships between different rainfall events and connectivity. Parsons (2012) also stresses that there is an urgent need for more emphasis on the timescale over which sediment moves through a catchment, specifically the rates of sediment movement of different sizes and from different sources.

At the catchment scale, SWAT seems to be efficient in representing the farm dams as a series of storages where connectivity is reduced. However, SWAT could not effectively identify wetlands as sink zones and simulations do not agree with the findings of Lorentz *et al*. (2011) that sediment originating from the cabbage plot is prone to be deposited in the wetland of sub-catchment 11. In contrast to farm dams, wetlands in SWAT are simulated off the main channel and water or sediment flowing into them must originate from the sub-catchment in which they are located (Neitsch *et al*., 2005). Therefore, it is recommended that future research in the Mkabela catchment include scenarios that account for wetland processes or impacts. For example, if the wetland is located alongside the channel the channel roughness coefficient in SWAT could be adjusted to represent wetland storage conditions. Long-term monitoring of discharge and sediment load is also recommended, including losses from evaporation and releases for irrigation in water bodies and sediment trap efficiencies.

In conclusion, SWAT results indicate the sensitivity of loads to hypothetical land use change, reflecting the spatial connectivity within the catchment due to the retention of loads mainly in the channel and farm dams. The study recommends that modellers should give sufficient attention to different connectivity aspects in sediment migration modelling, together with the way a model accounts for these aspects at different scales and from source to sink.

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4. METHODOLOGICAL FRAMEWORK FOR WATER EROSION RISK ASSESSMENT IN SOUTH AFRICA

Methodological framework

With the increase in human impacts on the environment, especially in terms of agricultural intensification and climate change, there is a need to accurately assess the distributed character of the erosion process and allow evaluation of at least the dominant contributing factors (Rubio and Bochet, 1998; Symeonakis Drake, 2004; Casalí *et al*., 2009). Advances in remote sensing, simulation models and GIS technology enable the processing of large amounts of data for application at a regional scale (De Jong *et al*., 1999; Wu and Wang, 2007). However, assessment at the regional scale is problematic due to spatial variability of the factors controlling erosion, the lack of input and validation data, as well as measurement variability (Zhang *et al*., 2002; De Vente and Poesen, 2005; Lenhart *et al*., 2005). Gullying and sediment movement are often ignored because of their variability at a regional scale (Boardman, 2006; Parsons, 2012).

Due to the fact that there are limitations to understanding each erosion process and scale at which assessment techniques can be applied (Drake *et al*., 1999), this study implemented a multi-process and multi-scale approach in Section 3. The approach includes assessment of (*i*) sheet-rill erosion at a national scale based on the principles and components defined in the (Revised) Universal Soil Loss Equation (R)USLE), (*ii*) gully erosion in a large catchment located in the Eastern Cape Province by integrating eleven important factors into a GIS, and (*iii*) sediment migration for a research catchment near Wartburg in KwaZulu-Natal by means of the Soil and Water Assessment Tool (SWAT). For each of these Case Studies, information is presented in the form of a methodological framework (Figure 1) encompassing different techniques and data to describe the main contributing factors and areas at risk. Figure 1 outlines the most feasible erosion assessment techniques and input datasets for which there is sufficient spatial information, emphasizing simplicity required for application at a regional scale with proper incorporation of the most important factors. The framework should not be interpreted as a single assessment technique but rather as an approach that guides the selection of appropriate techniques and datasets according to the complexity of the erosion processes and scale dependency. Several authors state that the selection of assessment techniques should be determined by the objective of the study, the scale, the dominant erosion processes and factors, as well as the availability of data (Boardman, 2006;

Vrieling, 2006; Van Zyl, 2007). In order to provide a comprehensive overview of the erosion risk, the framework illustrates that three hierarchical levels need to be implemented.

Hierarchical levels with increasing technique and data requirements

Three hierarchical levels for erosion risk assessment in SA, with increasing technique and data requirements, are illustrated in Figure 1. The first level allows for assessment of the spatial distribution of erosion risk and contributing factors at a national scale, emphasizing the sheet-rill aspects of the erosion process (Case Study *i*). This level should be followed by a second level that allows for assessment of the spatial distribution of gully erosion and contributing factors at a large catchment scale (Case Study *ii*). These levels provide no information about where material moves to or about connectivity between source and sink. A third level thus allows for assessment of sediment migration and associated connectivity aspects at a smaller/research catchment scale, including the influence of identified source and sink zones (Case Study *iii*).

The studies of Kirkby *et al*. (1996) and Drake *et al*. (1999) also suggested a hierarchical approach with three levels, although using different techniques at different scales compared to this study. Kirkby *et al*. (1996) assessed slope-scale water redistribution (first level), infiltration and vegetation interactions (second level), and soil aggregation (third level) in southeast Spain. Drake *et al*. (1999) conducted global scale modelling (first level), catchment scale modelling for the Walia catchment in Mali (second level), and proposed local scale assessment (third level) for areas that are identified as having accelerating erosion. Similar to the above-mentioned studies, nevertheless, the assessment techniques and data requirements used in this study increase with progression through the first to third level.

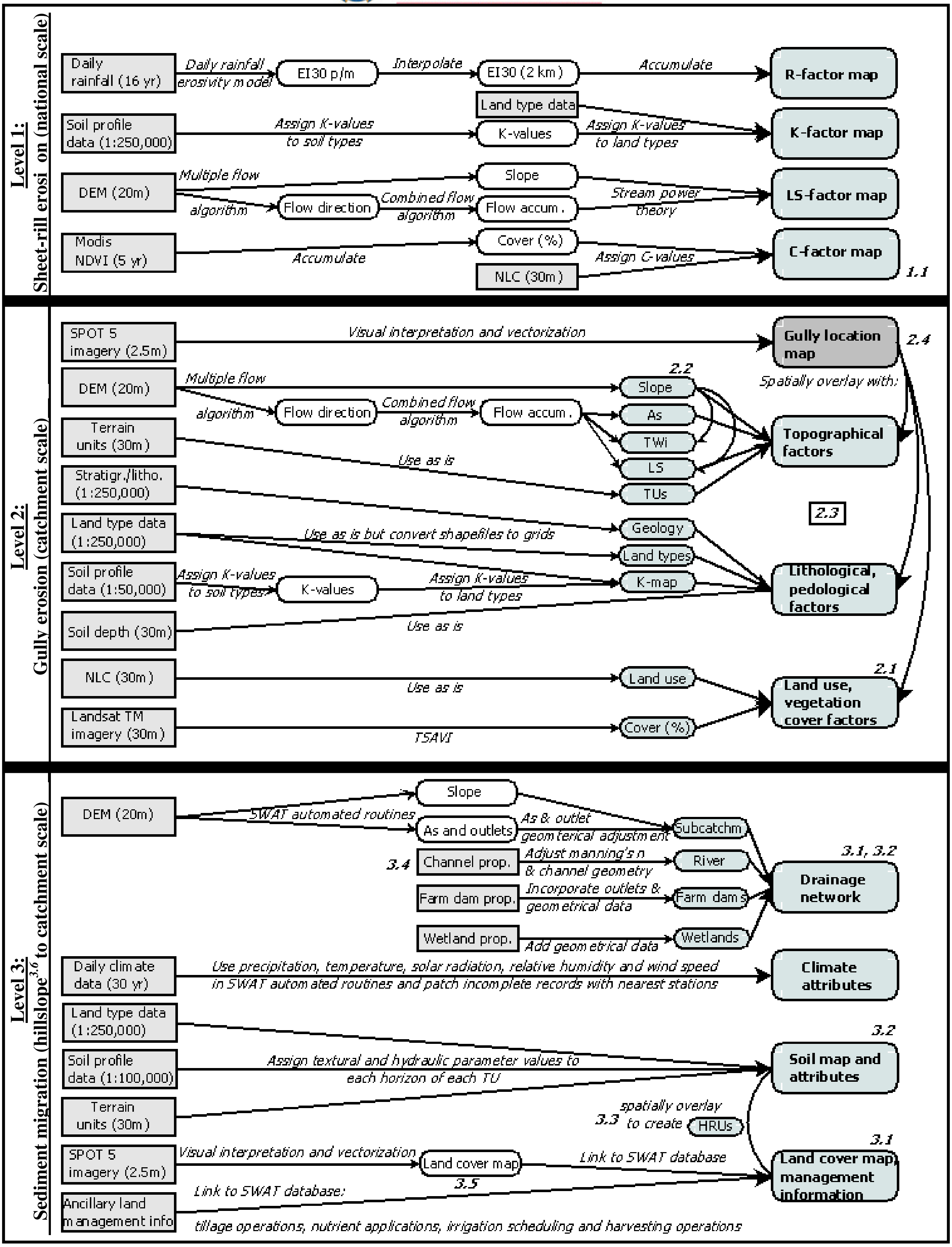

Figure 1: **Methodological framework for soil erosion risk assessment in South Africa (abbreviations and additional footnotes on following page).**

Figure 1 abbreviations:
As - upslope contributing area; C – Vegetation cover factor; DEM – digital elevation model; EI30 – Rainfall erosivity index; Flow accum. – flow accumulation; HRUs – hydrological response units; LS – topography factor or sediment transport capacity index; K – soil erodibility factor; MODIS - Moderate Resolution Imaging Spectroradiometer; NLC – National land cover; prop. – properties; R – rainfall erosivity factor; SPOT 5 - *Syste`me Pour l'Observation de la Terre*; Stratigr./Litho. – stratigraphic/lithologic; SWAT – Soil and Water Assessment Tool; TSAVI - Transformed Soil Adjusted Vegetation Index; TUs – terrain units; TWi - Topographic wetness index; Yr – year.

Figure 1 footnotes:
1.1 Combining the R-K- and LS-factors, and excluding the C-factor, provides the potential water erosion map of SA and represents the inherent susceptibility of the soil to erosion, irrespective of vegetation cover.
2.1 After quantifying the influence of factors in gully development, the identification of vegetated gully-free areas susceptible to gully development can be achieved by means of overlay analysis.
2.2 Each gully factor layer was categorized into 5 expert-based rankings or classes that, according to observations, uniquely influence gully development; due to the spatially thematic configuration of the gully factor layers it was decided to determine the proportion that each of the 5 classes are affected by gully erosion.
2.3 Although gully initiation occurs when certain rainfall and subsequent runoff thresholds are exceeded, this factor was not integrated in this analysis because threshold data were not available and the rainfall itself does not vary substantially in the central gullied part of the catchment.
2.4 Gully erosion rates can be modelled for representative test gullies and the results averaged over the areas of active gully erosion.
3.1 In order to create a final catchment overview of sediment migration downstream and associated connectivity aspects, the current study performed four scenarios: removal and expansion of the identified sediment source (cabbage plot) were performed to establish the extent that sediment outputs create input for downstream sub-catchments; whereas removal of the sediment sinks (9 farm dams and 5 wetlands) were performed to establish their impact on connectivity downstream.
3.2 Calibration should be achieved by adjusting the most sensitive model parameters. For example, the hydrological component can be calibrated by modifying the curve number and base-flow coefficients, whereas the erosion component can be calibrated by adjusting the soil erodibility and support management factors. Model performance can be improved by sequentially optimizing the widely used coefficient of efficiency of Nash and Sutcliffe (1970), as well as the coefficient of determination (r^2). As a measure of goodness-of-fit between simulated and observed loads, a simple per cent deviation method of Martinec and Rango (1989) can be used.
3.3 The overlay of land cover and soil maps creates hydrological response units (HRUs); portions of a sub-catchment that possess unique land use and soil attributes. Discretisation should be done to keep the number of HRUs down to a reasonable number, while considering the diversity and sensitivity of land cover and soil combinations.
3.4 Flow paths should represent all the relevant tributaries of the main river, whereas the unit links or sub-catchment outlets should spatially overlay with the flow monitoring points for calibration of model simulations with field measurements. In addition, channel erosion parameters can only be set to default representing non-erosive channels to eliminate channel erosion in simulations when all or most sediment is generated from agricultural fields.
3.5 The plant growth component of SWAT is a simplified version of the EPIC plant growth model (Sharpley and Williams, 1990), where phenological plant development is based on daily accumulated heat units developed by Monteith (1977) and biomass is inhibited by temperature, water or nutrient stress.
3.6. Although the importance of plot and hillslope scales is acknowledged (see for example Kakembo *et al*., 2012), it is not considered in the methodological framework.

In terms of remote sensing techniques, more sophisticated and/or time-consuming procedures such as pre-processing and specialized software are generally required to separate erosion features such as gullies from bare soil than automated procedures such as vegetation indices and classification techniques to create vegetal attributes. For example, automated techniques such as object-based modelling (i.e. eCognition® software and Erdas Imagine® Objectives Feature Extraction Model) cannot express individual gullies with the required accuracy over large areas due to their spectral complexity (Taruvinga, 2008; Pirie, 2009; Mararakanye and Le Roux, 2011). The imagery itself should have a pixel resolution smaller than the size of the erosion feature (Ludwig *et al*., 2007). Fortunately, with the development in sensor technology, space-borne data with improved spatial, spectral and temporal resolution is now available such as SPOT 5, IKONOS, Quickbird, WorldView and GeoEye (Smith and Pain, 2009). SPOT 5 satellite imagery was utilized in Case Study *ii* because the panchromatic sharpened images at 2.5 m resolution provide high resolution air photo-like quality for gully mapping (Taruvinga, 2008) and were acquired from government agencies for the whole of SA.

In terms of modelling techniques, more complex models are required to simulate sediment migration and associated connectivity aspects at the catchment scale than assessment of the spatial distribution of the erosion risk and contributing factors at a national scale. The main reason is that sediment migration is the integrated result of all erosion processes operating in a catchment (Vanmaercke *et al*., 2011; Parsons, 2012). Semi-distributed or semi-lumped models are often preferred above lumped models and fully-distributed or physically-based models, since the application of the former do not take connectivity aspects into account whereas the latter lead to additional errors and uncertainty resulting from more parameters and input data requirements in large catchments (Lenhart *et al*., 2005; Medeiros *et al*., 2010). The foundational strength of semi-distributed models such as SWAT is that it partitions the catchment of interest into homogeneous morphological units while considering most significant connectivity aspects, including factors controlling upland sediment generation, channel transport and deposition into sinks (Gassman *et al*., 2007). The combination of these models within a GIS framework is commonly utilized for erosion risk assessment (Flügel *et al*., 2003; Gao, 2008). GIS serves, not only as an output display platform, but also as a simulation interface and input data source. In addition to assessment techniques, Figure 1 also indicates regional datasets which are available for different erosion processes and contributing factors.

Similar to assessment techniques, data requirements increase with progression from the first to third level. More data are needed to simulate sediment migration at the catchment scale than erosion risk assessment and contributing factors at a national scale. According to Lenhart *et al*. (2005), this is the main reason sediment migration modelling has been restricted to applications in relatively small catchments for which high-quality data are available. A major limitation in the use of continuous time models such as SWAT in developing countries is the lack of recorded flow and sediment data for calibration and validation (Van Zyl, 2007; Boardman, 2006). Although higher accuracies are usually obtained when using high instead of low quality input data (De Vente *et al*., 2009) in many cases data with high spatial, spectral and temporal resolution do not exist for large areas (Van Rompaey *et al*., 2003). Besides, De Vente *et al*. (2009) found that differences in regional scale patterns are relatively small when using different input datasets. Furthermore, it is not feasible to collect input data required for existing models which exceeded data collection methods and compromises between data requirements and practical considerations are necessary (Collins and Walling, 2004).

Comparison between scales

In order to make the framework illustrated in Figure 1 more descriptive, important differences in technique and data requirements between national and catchment scale as well as the requirements for different processes, are highlighted for them. Here, the term catchment scale refers to both the large catchment and smaller research catchment utilized in Case studies *ii* and *iii* respectively. Figure 1 illustrates that assessment of the soil erosion risk at a national scale requires at least four main types of spatial input datasets including long term daily rainfall, soil data, digital elevation models and vegetation cover. These datasets are interfaced into a GIS and fed into appropriate and available algorithms to create soil erosion factor maps including rainfall erosivity, soil erodibility, topography and cover management factor maps. The product of these factor maps then provides the soil erosion risk map, also referred to as the Water Erosion Prediction Map of SA in Case Study *i* of Section 3.

At a catchment scale, more detailed processes need to be considered including gully erosion process (Van Zyl, 2007). Figure 1 further show that gully erosion mapping requires imagery with high spatial resolution, followed by the use of nine spatial input datasets to assess factor dominance. In order to assess factor dominance, the input datasets are integrated in a GIS to create eleven gully factor maps including topographical variables, parent material-soil associations and land use-cover interactions. Input data and assessment technique requirements for gully assessment at the catchment scale essentially double compared to national scale risk assessment. The main reason is assessment at a national scale does not purposefully target specific erosion processes but 'merges' them in a simplified empirical model, also referred to as a semi-quantitative model (De Vente and Poesen, 2005). Although the RUSLE model applied in this study emphasises the sheet-rill aspects of the erosion cycle by design, field observations indicate that most areas modelled as eroded on the risk map are recorded as having combinations of sheet-rill and gully erosion. In contrast, gully erosion risk assessment is explicitly aimed at separating gullies from other erosion processes, thereby accounting for the complexity associated with gully processes and contributing factors. Proper gully erosion risk assessment at a catchment scale generally involves more complex mapping and modelling techniques than soil erosion risk assessment at a national scale. Therefore, accurate gully erosion risk assessment at national scale would require spatial input data that are not currently available, as well as techniques that will not be feasible (too time-consuming). In addition to gully erosion, catchment scale assessment ideally/usually requires estimation of the migration of sediment from source to sinks (Parsons, 2012).

In addition to the input datasets required in levels 1 and 2, modelling the migration of sediment (level 3) at the catchment scale requires more quantitative hydrological data including water balance in the soil profile, hydrological structures and land management operations. Although this study used the SWAT model, these datasets could be utilized in any model that includes both a hydrological module and sediment module such as ACRU. As mentioned above, more complex models are required to simulate sediment migration and associated connectivity aspects at the catchment scale than assessment of the spatial distribution of the erosion risk and contributing factors at a national scale. The main reason is that sediment migration is the integrated result of all erosion processes operating in a catchment (Vanmaercke *et al.*, 2011; Parsons, 2012). Subsequently, assessment of sediment migration in catchments with gullies should not only include techniques and data described in level 3, but also the techniques and data described in level 2 or more.

Since data and technique requirements increase with progression through the first to third level, it is possible to apply the data and techniques of level 1 (national scale) at level 2 and 3 (catchment scale), but not the other way around simply due to the lack of data at a national scale and/or for most non-research catchments. Clearly, soil erosion risk assessment is very much data dependent, especially in developing countries such as SA. The following section briefly describes some important considerations and scale issues of the main factors controlling both sediment generation and migration at a regional scale.

Important considerations and scale issues

Since no study can incorporate the knowledge of all aspects of erosion, Gao (2008) states that it is important to understand to what spatial and temporal degree one needs to capture process dynamics for the purpose of the study, and to apply the most appropriate and practical technique. Soil erosion risk assessment techniques should be adapted and modified to combine sufficient simplicity for application at a regional scale with proper incorporation of the most important contributing factors (Gobin *et al*., 2003). However, contributing factors can only be assessed at a regional scale if they and their scale issues are understood in a geographical context (Molenaar, 1996). This section describes some important considerations and scale issues of the dominant factors in order to guide and facilitate standardization of future regional assessment efforts in SA. The scale issues referred to here relate to changes in the methods or resolution used for data collection in the three Case Studies of Section 3 and not to those concerning upscaling or downscaling of erosion processes. The main factors contributing to sediment generation and migration at a regional scale include rainfall, parent material-soil associations, topographic-drainage-network variables, and land use-cover interactions.

Rainfall

Sediment generation and transport largely depends on rainfall duration and intensity (Bracken and Croke, 2007). Unfortunately, rainfall intensity data are usually incomplete and/or have short recorded periods at a regional scale particularly in developing countries. The best alternative is to use daily rainfall data in empirical relationships between rainfall intensity data and daily rainfall amount. Care needs to be taken to insure that the rainfall erosivity algorithms used are not solely a function of rainfall amount. Irrespective of the rainfall amount, winter rainfall produced by frontal activity is less erosive compared to thunderstorms associated with convection during summer in SA. Case Study *iii* in Section 3 clearly illustrate a summer dominant erosion pattern in the Mkabela catchment which is mainly caused by intensive summer rainfall events, where the infiltration capacity of the soil is exceeded. Laker (2004) also states that the episodic nature of rainfall in SA can be a strong control on soil loss rates. Extreme events and seasons produce the most erosion and runoff when break-of-season rains following long periods of drought (see also Hamblin, 2001). Vegetation cover is severely denuded during prolonged droughts, leaving the bare soil exposed to torrential rains that often follow (Snyman, 1999).

Selection of an erosivity algorithm should thus consider the climatic variations and conditions of intended use. The period of interest must accommodate natural climatic variations and include a variety of climatic conditions (above-normal rainfall and drought). For example, Case Study *i* in Section 3 used the daily rainfall erosivity algorithm developed by Yu and Rosewell (1996a and 1996b) in Australia for two main reasons (Le Roux *et al*., 2008). First, Australia has a climate that, similar to SA, ranges spatially between winter rainfall areas in the southwest to a summer rainfall with tropical influences over the northern parts. Second, large areas over the interior of both countries are classified as semi-arid while coastal regions in the east receive high rainfall amounts. It is further recommended to assess the impact of rainfall with relation to vegetation cover on soil erosion (see Lu *et al*., 2003). Laker (2004) stresses that some areas in SA with high rainfall are inversely related to soil loss due to good vegetation cover and stable soils from advanced pedogenesis.

Parent material-soil associations

Several authors state the importance of soil as an erosion factor in SA (e.g. Beckedahl, 1998; Rienks *et al*., 2000; Laker, 2004; Le Roux and Sumner, 2012). Although the physical, as well as chemical, soil properties and their interactions that affect soil erodibility are many and varied, most models focus on topsoil properties such as texture and structure. Coarse textured soils with a strong structure (fine granular) render the soil resistant to detachment and have low erodibility values, whereas fine textured soils with low-density aggregates (blocky, platy or massive structure) are carried more easily by overland flow and have high erodibility values. Some properties that influence soil erodibility in SA, however, do not feature in (R)USLE type models. Therefore, in all three Case Studies a modified version of the Soil Loss Estimator of Southern Africa (SLEMSA) model was used as a guide to the assignment of (R)USLE soil erodibility factors to the land types of SA (Land Type Survey Staff, 1972-2006). SLEMSA was chosen because it was developed for use in southern Africa and particularly for the manner in which topsoil and subsoil structure are incorporated (see Appendix A).

Nevertheless, some of the most important hydraulic (available water capacity and saturated hydraulic conductivity) and chemical (organic matter content, free iron oxides, Mg:Ca ratios, sodium exchangeability and clay mineralogy) parameters could not be quantified or modelled in any of the three Case Studies due to the limited range of descriptive soil information available at a regional scale. Soil dispersibility is probably the most important soil property that could not be analyzed by the Case Studies because differences are too large between soil types. For example, relationships between sodium exchangeability and crusting are region specific and threshold values can only be drawn if they are determined separately for different groups of soils with similar clay mineralogy and/or geology (see D'Huyvetter, 1985; Bloem and Laker, 1992; Bühmann *et al*., 1996).

Several authors state the importance of parent material in terms of soil erodibility (e.g. Dardis *et al*., 1988; Watson and Ramokgopa, 1997; Le Roux *et al*., 2008). However, eroded soils do not always, or simply, correlate spatially with weak underlying geology. As mentioned in Case Study *ii*, one would expect to find higher proportions of gullies in transported/unconsolidated alluvium and weak sedimentary mudrock that usually give rise to deep erodible soils (Laker, 2004). The most probable reason for latter discrepancy is that quantification of factor dominance is complicated by the relatively large spatial extent of stratigraphic polygons (aggregated geological types) as described by the Council for Geoscience (2007); not because of the lack of geological variability as indicated in several other studies (Verbist *et al*., 2010). Another reason that gullied soils do not always, or simply, correlate spatially with weak underlying geology is that gully development is enhanced by other factors.

Topographic-drainage-network variables

The three Case Studies in Section 3 indicate that topographic factors and/or drainage networks should be constructed in order to represent the movement of runoff and sediment downslope from hydrologically active areas to stream channels and further downstream. Most studies agree that topographical parameters should be determined from fine resolution digital elevation models (DEMs) (e.g. <30 m) resulting in computed topographic surfaces with less variance and uncertainty than coarse resolution DEMs (>30 m) (see e.g. Mitasova *et al*., 1996; Gertner *et al*., 2002; De Vente *et al*., 2009). The main reason that resolution is important is tied to the spatial variability of topographic factors, influencing processes at a fine spatial scale. Coarse DEMs tend to have a “smoothing” effect on computed topographic surfaces. High altitude areas are lower whereas low altitude areas are higher and short steep slopes tend to disappear, reducing the resultant slope estimate and insinuating a higher connectivity (Zhang *et al*., 2002; Verstraeten, 2006; De Vente *et al*., 2009). The finest resolution DEM used in all three Case Studies available in SA at a national scale is a DEM interpolated from contour data by GISCOE (2001) with a grid cell size of 20 m. However, when using this DEM users should be cautious of artificial pits/sinks, especially in flat areas, because the DEM is not hydrologically corrected such as the improved but still coarse Shuttle Radar Topography Mission (SRTM) DEM at 90 m resolution (Weepener *et al*., 2011).

Automated procedures are required to determine topographical variables for complex terrain at a regional scale. Extraction of stream networks or flow path lines in the Case Studies in Section 3 are therefore conducted by algorithms that accumulate the contributing area upslope of each pixel through a network of cell-to-cell drainage paths (Band and Moore, 1995; Gallant and Wilson, 2000). Combined flow algorithms are recommended since they simulate more realistic flow networks by combining multiple and single flow procedures to represent flow dispersion in upland areas, as well as channel convergence further downslope respectively (Freeman, 1991; Wilson and Gallant, 2000). In addition to the flow algorithms, a variety of models exist that connect sediment sources with the river channel and further downstream (Lenhart *et al.*, 2005).

Case Study *iii* used the SWAT model, a semi-distributed or semi-lumped model that partition the catchment of interest into homogeneous morphological units allowing to certain extents the spatial variation to be accounted for (see also Lenhart *et al.*, 2005; Gassman *et al.*, 2007). Semi-distributed or semi-lumped models are often preferred above fully-distributed or physically-based models, since the application of the latter in large catchments lead to additional errors and uncertainty resulting from more parameters and input data requirements (Medeiros *et al.*, 2010). When using semi-distributed models, however, care is needed in selecting unit sizes so that spatially aggregated areas adequately represent the spatial variability in the catchment. Importantly, the flow paths should represent all the relevant tributaries of the main river, whereas the unit links or sub-catchment outlets should spatially overlay with the flow monitoring points for calibration of model simulations with field measurements. According to observations in the field, channels should be subdivided into segments with unique geometric (slope, length and width) and roughness (e.g. Manning's roughness coefficient) properties.

In addition, stream channel processes and hydrological structures need to be characterised, allowing deposition of excess sediment depending on the carrying capacity and/or sediment storages where connectivity is reduced (Chen and Mackay, 2004). Case Study *iii* illustrates that farm dams within the Mkabela catchment are particularly efficient storages where flow speed is reduced and sediment deposited. Several other studies indicate that the effect of sediment sinks become dominant over sediment sources with increasing spatial scale (Kirkby *et al.*, 1996; De Vente and Poesen, 2005; De Vente *et al.*, 2007; Lesschen *et al.*, 2009). It is therefore not surprising that, within the Mkabela catchment, changes in sediment production upstream have no significant changes on the sediment yield at the catchment outlet (see also Walling, 1999; Prosser *et al.*, 2001). The reduction in connectivity with increasing spatial scale or catchment area (> ~10 km^2) is a globally recognized trend although this varies regionally (De Vente *et al.*, 2007). Sediment yield can increase or decrease at any catchment area due to the spatial variability of the factors influencing soil erosion and sediment yield, such as land use-cover interactions (De Vente *et al.*, 2007).

The advantages in applying widely-used approaches in the Case Studies in Section 3 were offset by a few disadvantages. The specifications or input values including slope exponents, flow accumulation/slope-length threshold values and maximum cross grading area used for the calculation of slope-length, were based on values recommended in other sources of literature (Freeman, 1991; Renard *et al*., 1994; Wilson and Gallant, 2000; Schäuble, 2003). However, optimum values depend on local conditions and the use of reference parameter values over large areas may lead to errors. (R)USLE based studies tend to overestimate erosion rates in areas with steep terrain, for example along the escarpment in SA (Le Roux *et al*., 2008), especially since (R)USLE was developed in the US where topographic features are considered to be a dominant factor (Laker, 2004). Collectively, the factors discussed above highlight areas that are intrinsically susceptible to erosion and sediment yield. The last parameter discussed below is important in highlighting areas where erosion is extrinsically triggered or accelerated by land use and human-induced reduction of the vegetation cover.

Land use-cover interactions

It is generally agreed that land use and vegetation cover interactions are dominant above all factors. Vegetation indexes such as the Normalised Difference Vegetation Index (NDVI) are an important source of information for vegetation cover at a regional scale (Gobin *et al*., 2003). However, NDVI data are sometimes inaccurate due to the effect of soil reflectance and the sensitivity to vitality of the vegetation. In Case Study *i*, the fractional vegetation cover estimated for Fynbos in the Western Cape is probably too low, leading to over-estimated soil-erosion values. This problem occurs during vegetation senescence when vegetation indices decrease even when cover remains the same, but senescent vegetation offers the same protection to soil as green vegetation and it is also important to detect relatively dry vegetation (see e.g. French *et al*., 2000; Odindi and Kakembo, 2009). It is therefore recommended to use imagery that depicts conditions for which differentiation is easily obtained between green vegetation and bare soil, as opposed to dry vegetation which is more difficult to detect. It is also recommended to use soil adjusted vegetation indices such as the Transformed Soil Adjusted Vegetation Index (TSAVI). Case Study *ii* used TSAVI because it leads to a significant reduction of the soil effects for areas of sparse vegetation or bare soil and delivers reliable results for arid and semi-arid grassveld landscapes in SA (see also Flügel *et al*., 2003; Hochschild *et al*., 2003). Nevertheless, soil adjusted indices have difficulty in accounting for spatially variable soil types, especially at a regional scale.

Apart from the canopy cover, ground cover is not always represented in remotely sensed data. For example, in Case Study *i* the fractional vegetation cover for savanna in northern

Limpopo and Northern Cape remain questionable due to the dense tree canopy concealing the poor ground cover when monitored by satellite. Field observations indicate that arid area ground cover is frequently less than its projected vegetation crown cover, which is not always protective against erosion. Soil is especially susceptible to erosion when the ground cover is at its lowest and rainfall erosivity at its highest. Although it is recommended to use multi-temporal satellite imagery to account for the interaction between vegetation growth and rainfall (see Lu and Yu, 2002; Lu *et al*., 2003), multi-temporal analysis is beyond the scope of this research.

In terms of spatial scale, results depend on the grid-cell resolution since land use and vegetation parameters are carried out in a grid-based system (Zhang *et al*., 2002). Zhang *et al*. (2002) illustrate how grid-cell resolution affects predicted erosion. Coarse resolution grids reduce predicted erosion due to bare soil areas being incorrectly imbedded in vegetated areas. This explains why more vegetation is illustrated in discontinuous gullies than continuous gullies (Case Study *ii* in Section 3). The Landsat TM image used to calculate the TSAVI and subsequent cover grid have a coarse resolution of 30 m^2 and therefore, small gullies with narrow patches of bare soil are incorrectly imbedded in vegetated areas (see also Taruvinga, 2008). Since discontinuous gullies are frequently less than 30 m^2 in size, the proportion of vegetation cover inside these gullies could be overestimated, while the proportion of bare soil could be underestimated. In addition to spatial resolution, Case Study *iii* indicates that the location and organization of land cover pixels is also important, especially in terms of sediment transport and connectivity between sources and sinks (see also Van Rompaey *et al*., 1999; Van Oost *et al*., 2000; Kakembo *et al*., 2012). However, a major weakness of SWAT is that it does not consider the processes of deposition during transport from hillslopes/HRUs to channel (Lenhart *et al*., 2005). The cabbage plot sediment is not spatially identified within the sub-catchment it is located and all the sediment generated from the cabbage plot reaches the channel, whether connected to the channel or not. Chen and Mackay (2004) suggest the use of smaller sub-catchments instead of HRUs, but this approach is not applicable in relatively large catchments such as simulated in Case Study *iii*. In such catchments, discretisation should be done to keep the number of sub-catchments and HRUs down to a reasonable number, while considering the diversity and sensitivity of land cover and soil combinations. It is imperative that the spatial resolution and organization adequately represent the spatial variability in the catchment and that all the significant land cover and soil combinations affecting sediment yield are integrated.

In addition to spatial resolution, the imagery itself needs to consist of an adequate spectral resolution. When using remotely sensed images for detecting erosion features amongst vegetation and bare soil, it is important to consider which imagery will provide the most appropriate information. For example, the spatial resolution (2.5 m panchromatic band) of SPOT 5 imagery used in Case Study *ii* were adequate to manually vectorize gullies in a large catchment, but automated mapping techniques could not express individual gullies with the required accuracy due to the imagery's limited spectral resolution of only 3 bands. The spectral reflectance between gullies varies significantly and depends on vegetation cover inside gullies, as well as several soil properties such as the soil organic matter and soil moisture contents. In order to account for the spectral variability of South African landscapes at a regional scale, sophisticated and time-consuming strategies such as spectral unmixing are required for endmember selection that are outside the scope of text (see e.g. Haboudane *et al.*, 2002).

Lastly, land use and management information are also important including tillage operations, nutrient applications, irrigation scheduling, harvesting operations and support practices. Of all the erosion factors, however, this factor or set of parameters are the least reliable due to the lack of data on crop rotation systems and timing of agricultural operations at a regional scale (Renard *et al.*, 1994). The most practical way to incorporate this information is to link a land cover map to an existing model structure and database consisting of several plant growth parameters where phenological plant development is based on daily accumulated heat units. In Case Study *iii*, for example, the land cover map was digitized from SPOT 5 imagery acquired in 2006 and linked to the plant growth component of the SWAT model. SWAT utilizes a simplified version of the EPIC plant growth model (Sharpley and Williams, 1990), where phenological plant development is based on daily accumulated heat units developed by Monteith (1977) and biomass is inhibited by temperature, water or nutrient stress. Despite a lack of reliability and associated uncertainties of such data and the subsequent modelled outputs, these parameters are specifically useful to determine the relative impact of different land use and management scenarios, as well as for comparative purposes with possible future scenarios under land use and climate change.

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5. CONCLUSIONS AND RECOMMENDATIONS

With the increase in human impacts on the environment, especially in terms of agricultural intensification and climate change, there is a need to assess and continually monitor the erosion process and contributing factors (Rubio and Bochet, 1998; Symeonakis Drake, 2004; Casalí *et al*., 2009). SOILOSS (Lu *et al*., 2003), BASINS (US EPA) and PESERA (Kirkby *et al*., 2004) are standardized approaches that assess the soil erosion problem according to conditions in Australia, the U.S.A. and Europe respectively. In most other countries, however, especially in developing countries such as South Africa (SA), there is still an absence of standardized methodological frameworks that deliver comparable results across large areas as a baseline for regional scale monitoring. Although a fair number of attempts have been made to assess the soil erosion risk in SA (described in Section 2), there is hitherto no practical methodological framework to provide a consistent baseline for regional scale monitoring under South African conditions.

Since there are limitations to understanding each erosion process and scale at which assessment techniques can be applied (Drake *et al*., 1999), this study implemented a multi-process and -scale approach using different techniques and data in order to assess different soil erosion processes and contributing factors. These include assessment of (*i*) sheet-rill erosion at a national scale, (*ii*) gully erosion in a large catchment and (*iii*) sediment migration for a smaller research catchment. More specifically, the first Case Study presented in Section 3 based sheet-rill erosion prediction on the principles and components defined in the (Revised) Universal Soil Loss Equation (R)USLE (Renard *et al*., 1994). These results are followed by a description of the limitations and challenges that must be overcome in soil-erosion assessment on a national scale. The second Case Study presented in Section 3 assessed gully factor dominance in a large catchment located in the Eastern Cape Province by integrating eleven important factors into a GIS that could be readily derived for the whole area. The study also highlights gully factors likely to emerge as dominant between continuous gullies and discontinuous gullies. The third Case Study utilized the frequently applied Soil and Water Assessment Tool (SWAT) (Arnold *et al*., 1998) to assess sediment migration and associated connectivity aspects in a catchment (Mkabela near Wartburg, South Africa) with identified source and sink zones. Insight is also provided into the applicability of SWAT in connectivity studies, explicitly describing how model assumptions affect outputs in context of connectivity between sources and sinks. The following conclusions can be drawn from the study.

Results presented in the Case Studies of Section 3 re-enforce that soil erosion is a major soil degradation problem in South Africa. Case Study *i* illustrates that approximately 50% (61 million ha) of national land has a moderate to severe erosion potential (>12 t/ha·yr), whereas approximately 20% (26 million ha) of land is classified as having a moderate to severe actual erosion risk. The Eastern Cape Province makes the largest (28%) contribution to soil loss with approximately one third (16 million ha, 37%) of the province classified as moderate to extremely high. Case Study *ii* identifies severe gully erosion, affecting an area of approximately 5 273 ha in the large catchment (the Tsitsa valley) of the Eastern Cape Province. Case Study *iii* illustrates that a cabbage plot in the upper reaches of the research catchment near Wartburg is a significant sediment source, but is counterbalanced by sinks including the river channel and farm dams downstream.

The factor-based nature of this multi-process and -scale approach allowed scrutiny of the role of the main factors in contributing to the erosion risk. In terms of sediment generation (sheet-rill and gully in Case Studies *i* and *ii*), it appears that the combination of poor vegetation cover and susceptible parent material-soil associations are the overriding factors in South Africa, and not the topography and rainfall as frequently determined in the USA and Europe (e.g. Vanmaercke *et al*., 2011). Several studies in other parts of the world found similar results that parent material-soil associations and land use-cover interactions control erosion risk at a regional scale. For example, Tamene *et al*. (2006) determined that the major controlling factors in a mountainous dryland region of northern Ethiopia include surface lithology and land cover. Topography, sometimes together with soil type and rainfall, seems to play a more important role at a local scale (Lane *et al*., 1997; Verstraeten, 2006; Bracken and Croke, 2007) but their assessment and influence should not be ignored due to high levels of accuracy required for policy and management purposes. In terms of sediment migration, Case Study *iii* compares well with the global trend that sediment sinks becomes dominant over sediment sources with increasing spatial scale (see Kirkby *et al*., 1996; De Vente and Poesen, 2005; De Vente *et al*., 2007; Lesschen *et al*., 2009). Hydrological structures such as farm dams can be particularly efficient storages where connectivity is reduced and sediment deposited (Mishra *et al*., 2007). Nevertheless, it is recognised that the reduction in connectivity with increasing spatial scale or catchment area varies regionally (De Vente *et al*., 2007). According to De Vente and Poesen (2005), not all processes and contributing factors are equally important in different regions. Hence, it is important to understand to what spatial and temporal degree one needs to capture process dynamics for the purpose of the study, and to apply the most appropriate and practical technique (Gao, 2008).

Section 4 presents the methodological framework illustrating the most feasible erosion assessment techniques and input datasets for which sufficient spatial information exists, and emphasizes simplicity required for application at a regional scale with proper incorporation of the most important factors. In order to provide a comprehensive overview of erosion risk, the framework illustrates that a multi-process and -scale approach with three hierarchical levels needs to be implemented (see simplified version of the framework in Figure 1). The first level allows for the assessment of the spatial distribution of erosion risk and contributing factors at a national scale, emphasizing the sheet-rill aspects of the erosion process. This level should be followed by a second level that allows assessment of the spatial distribution of gully erosion and contributing factors at a large catchment scale. These levels provide no information about where material moves to or about connectivity between source and sink. A third level allows for assessment of sediment migration and associated connectivity aspects at a smaller/research catchment scale, including the influence of identified source and sink zones. As mentioned above, Kirkby *et al.* (1996) and Drake *et al.* (1999) also applied/suggested a hierarchical approach with three levels, but using different techniques in different areas at finer or coarser scales than applied in this study.

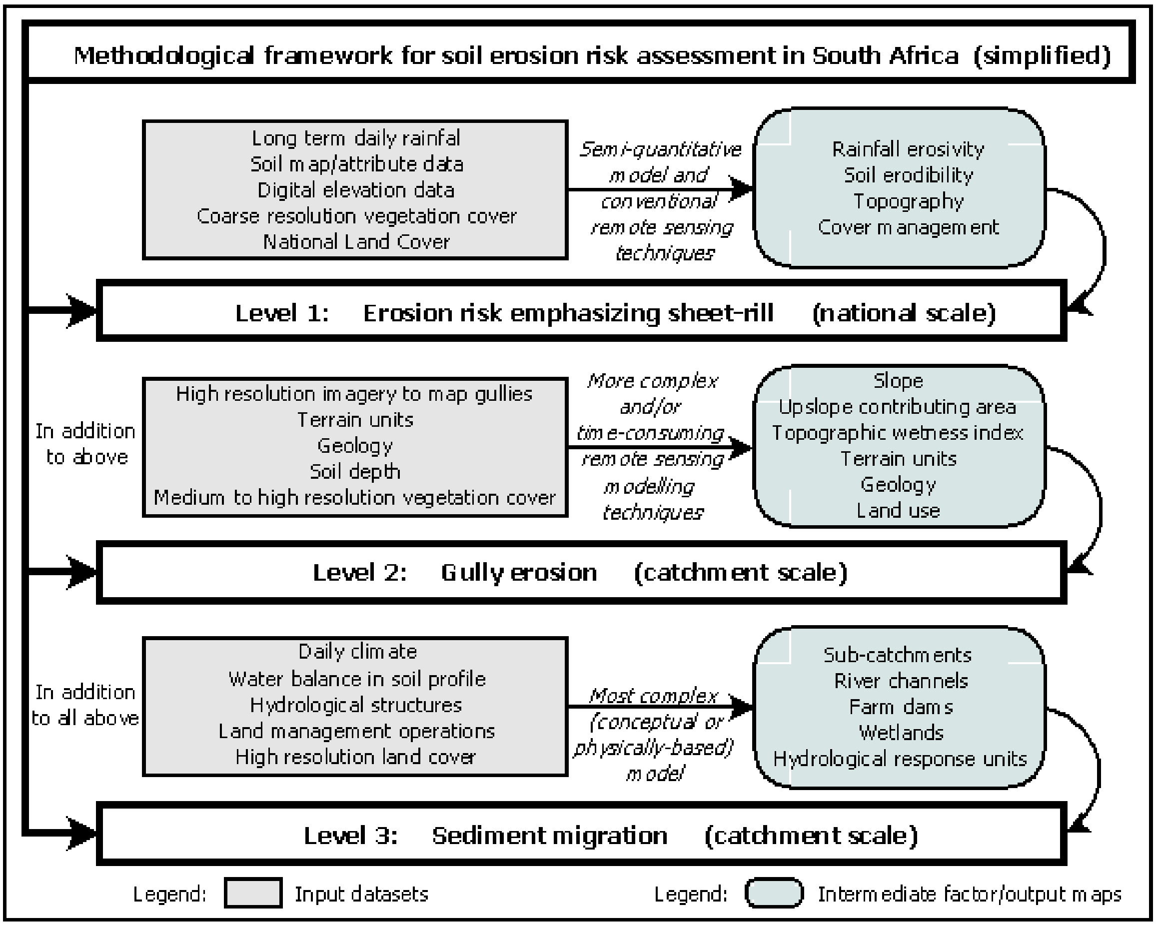

Figure 1: Methodological framework for soil erosion risk assessment in South Africa (simplified).

Assessment techniques and data requirements increase with progression through the first to third level. In terms of remote sensing techniques, more sophisticated and/or time-consuming procedures such as pre-processing and specialized software are required to separate erosion features such as gullies from bare soil than automated procedures such as vegetation indices and classification techniques to create vegetal attributes. In terms of modelling techniques, more complex models are required to simulate sediment migration and associated connectivity aspects at the catchment scale than assessment of the spatial distribution of the erosion risk and contributing factors at a national scale. Similar to the assessment techniques, data requirements increase with progression through the first to third level; more data are generally needed to simulate sediment migration at the catchment scale than erosion risk assessment and contributing factors at a national scale. Since data and technique requirements increase with progression through the first to third level, it is possible to apply the data and techniques of level 1 (national scale) at level 2 and 3 (catchment scale), but not the other way around simply due to the lack of data at a national scale. Clearly, soil erosion risk assessment is very much data dependent, especially in developing countries such as South Africa.

Further refinement of soil erosion risk assessment will be possible given additional research, including the following. It is recommended to consider all sediment yielding processes and assess the sediment supply from each. In addition to water erosion, Symeonakis and Drake (2004) state that the effects of wind need to be considered, especially in windy plains with fine sandy soils such as in some areas in the Kalahari and Karoo. According to Van Zyl (2007), the ability to account for sediment supply from gullies will be an important feature in catchments with gullies as the predominant source of sediment. Gully erosion rates can be modelled for representative test gullies and the results averaged over the areas of active gully erosion (Flügel *et al.*, 2003). Furthermore, the effect of socio-economic drivers of erosion risk needs to be investigated. Boardman (2006) states that farmers are too often not considered in studies but the way farmers perceive their role influences their attitude to erosion and conservation.

Multi-temporal analysis should be used to investigate the effect of land use history and vegetation conditions (e.g. Kakembo *et al.*, 2009; Kakembo *et al.*, 2012) and can also be used to account for the interaction between vegetation growth and rainfall (Lu and Yu, 2002; Lu *et al.*, 2003). Another example where multi-temporal analysis is required is to distinguish between active and passive erosion features. Active gully erosion features are of major importance, because large active gullies usually deliver the main quantity of sediments in a catchment, whereas, stable gullies have little or no sediment delivery. Parsons (2012) stresses that adequate consideration should be given to the time taken for sediment to travel via the various pathways to the catchment outlet, specifically the rates of sediment movement of different sizes and from different sources.

It is also recommended to quantify the individual factor dominance of the main factors contributing to each erosion process. Although the factor-based nature of this multi-process and -scale approach allowed scrutiny of the role of the main factors in contributing to the erosion risk, this study did not quantify factor dominance contributing to each erosion process. Factor dominance was only quantified in Case Study *ii* for gully erosion in a large catchment by integrating the factors into a GIS. According to Wu and Wang (2007), contributing factors may differ between different regions and their individual degrees of risk and corresponding comparative weights may call for re-evaluation.

It is further recommended to increase efforts of continuous long-term monitoring of discharge and sediment load in South Africa, including reservoir sedimentation and trap efficiencies. There is a need for datasets comprising spatially distributed data of recorded flow and sedimentation, especially for calibration and validation (see also Van Zyl, 2007; Boardman, 2006). Calibration and verification of results should preferably consider the spatial distribution and absolute values if possible including flow characteristics, sediment generation at source areas, as well as sediment transport and deposition exhibited in sinks. Sediment fingerprinting best represents the direct approach to sediment sourcing and offer considerable potential for meeting this requirement (Collins and Walling, 2004). However, tradeoffs are necessary between the extra effort and increase in accuracy of results (De Vente and Poesen, 2005).

Despite the lack of reliability and associated uncertainties of data and subsequent modelled outputs, the methodological framework presented here provides descriptions of the contributing factors for standardized definition of the soil erosion risk in South Africa. In conclusion, the framework outlines the techniques that should be used and the data that are available to identify areas at risk. The framework should not be interpreted as a single assessment technique but rather as an approach that guides the selection of appropriate techniques and datasets according to the complexity of the erosion processes and scale dependency. A set of guidelines (important considerations and scale issues) are provided in order to allow evaluation of at least the dominant factors that contribute to different processes. In addition, in future it will be useful to determine the relative impact of different land use and management scenarios, as well as for comparative purposes with possible scenarios under climate change. It is often argued that climate change will increase future erosion rates, especially where increased rainfall intensity and/or extreme event frequency are predicted (Boardman, 2006). However, Boardman (2006) stresses that certain land use changes causing a reduction in the vegetation cover are likely to have greater impact on the erosion risk than any likely climate change. Therefore, the vegetated erosion-free areas susceptible to erosion that were identified in this study are important in policy terms. Results indicate those areas which are inherently susceptible to erosion, but which are presently protected by vegetation. Appropriate strategies need to be designed for these susceptible areas in order to protect the current vegetation cover.

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APPENDIX A:

MAPS OF FACTORS INFLUENCING SHEET-RILL EROSION AT A NATIONAL SCALE

The annual erosivity map shows an east to west gradient over SA with low (100-300 MJ.mm/ha.hr.yr) erosivity values over the dry western parts of the country and high (20 000-40 000 MJ.mm/ha.hr.yr) erosivity values over the eastern parts of the country (see Figure 1). Over the southwestern Cape erosivity values are lower than over the summer rainfall areas where similar annual rainfall occurs. Therefore, the model correctly compensated for lower rainfall intensities over the winter rainfall areas and higher intensities over the summer rainfall areas. While winter rainfall produced by frontal activity is of a more general and light nature, thunderstorms associated with convection during summer produce higher rainfall intensities. The highest erosivity values occur along the escarpment, especially northward, where the influence of tropical low pressure systems from time to time cause heavy rainfall and very high daily rainfall totals. Erosivity values calculated over mountainous areas are two to three times higher than those previously calculated by Smithen and Schulze (1982). This is the result of more stations used as well as the inverse distance weight method of interpolation that better compensates for topographical influences.

Figure 2 indicate that high to very high soil erodibility values in range of 0.022 - 0.046 t/ha/(MJ.mm/ha.hr) are found in a number of regions in SA, some of the most prominent being in the southern Free State, as well as the northern and southern regions of the Eastern Cape. In terms of texture, soils with high clay content usually have low k-factor values because they are generally resistant to detachment with strong cohesion between the clay particles (e.g. Shortlands along the Lebombo mountain range). Soils with a high permeability prevent runoff and erosion, and therefore generally have low K-factor values (e.g. coarse sandy soils of the Kalahari Desert). In terms of structure, transported sediment and unconsolidated soil with a Neocutanic horizon, usually have high K-factor values because they are easily detached and transported by overland flow (e.g. transported colluvial and alluvial sediments of the Mtata River in the Eastern Cape) (Tooth *et al.*, 2000). Orthic topsoils often have high k-factor values due to a weak structure caused by wetness or waterlogging (e.g. Kroonstad Katspruit form). Soils with an E-horizon are also weakly structured or structureless and erodible due to periodic saturation with water and *in situ* removal of colloidal cementing matter including clay, iron oxides and organic matter (e.g. Fernwood near Humansdorp) (MacVicar *et al.*, 1977). The removal of colloidal matter is also the reason why soils with a clear transition from overlying horizons are erodible (e.g. Swartland form near Stanger) (Fey, 2010). In many cases soils have an abrupt transition between the topsoil and the subsoil with respect to texture, structure and consistence (e.g. Sterkspruit Duplex soils). These soils are highly erodible due to a permeable horizon overlying abruptly a less permeable one, causing water to infiltrate and saturate the top layer where it moves in a predominantly lateral direction as subsurface flow (MacVicar *et al*., 1977). Finally, in terms of soil depth, deep soils usually have low K-factor values because they have higher water-holding capacities and are able to absorb larger rainfall amounts before overland flow is generated, whereas shallow soils with minimal development and lithic contact on steep slopes have high K-factor values (e.g. Mispah and Glenrosa soil forms between Douglas and Vryburg in the Northern Cape) (Samadi *et al*., 2005).

The LS-factor map is shown in Figure 3. Results illustrate that high LS-factor values follow the topography, especially in the escarpment. Long steep slopes, a common feature in the KwaZulu-Natal Drakensberg, render the land extremely susceptible to erosion (Schulze, 1979). Other areas of pronounced relief include large tracts of the former Transkei and Waterberg Plateau. It is worth mentioning here that not all these areas are necessarily affected by high erosion rates. Some areas have a high potential erosion risk but a low actual erosion risk due to good vegetation cover and/or stable soils. The problem is that (R)USLE based studies tend to overestimate erosion rates in areas with steep terrain (e.g. along the escarpment in SA), especially since (R)USLE was developed in the US where LS features very prominently and is considered to be a dominant factor (Laker, 2004).

The C-factor map is illustrated in Figure 4. The C-factor map indicates that the highest C values are ascribed to the western and northern arid parts of SA (0.6). The eastern marginal zone of SA (approximately 42 million ha) positioned between the interior plateau and coast (0 – 1200 m a.s.l.) has the lowest C-values (0.003). Low C-values (good cover) in the eastern marginal zone are essential to compensate for the high potential erosion risk and it is recognised that there is a huge difference between actual and potential soil erosion for this region.

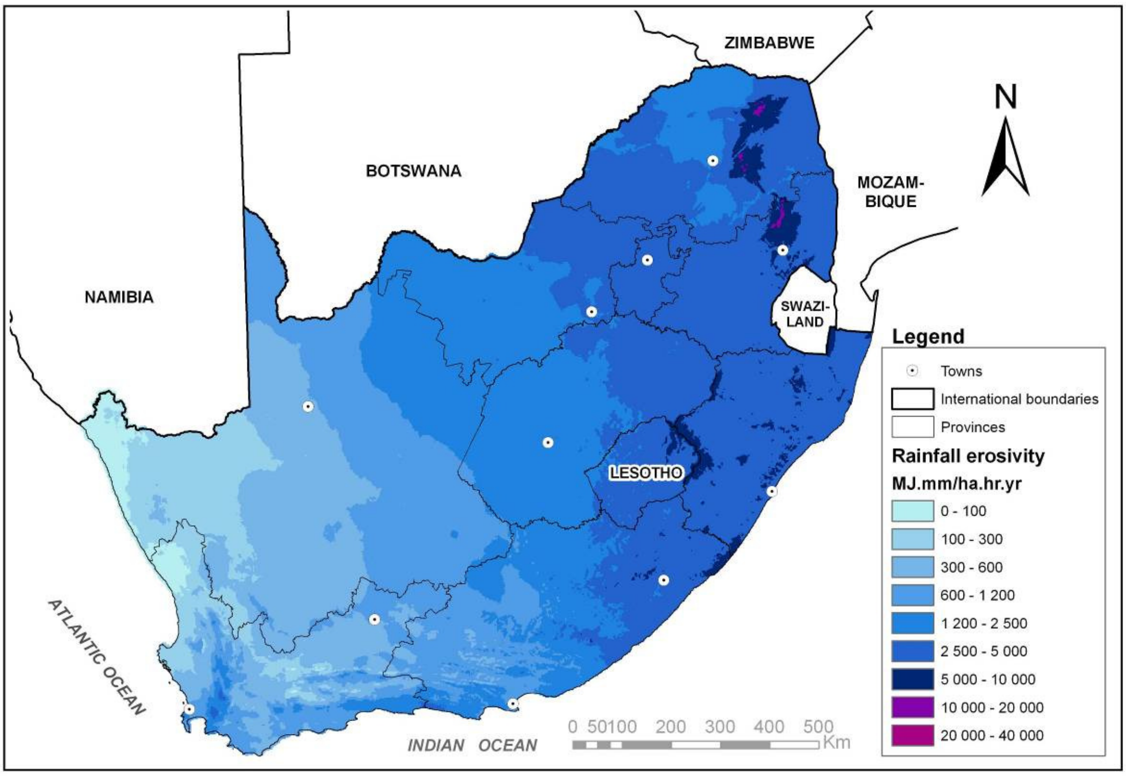

Figure 1: Rainfall erosivity factor (R) map of South Africa.

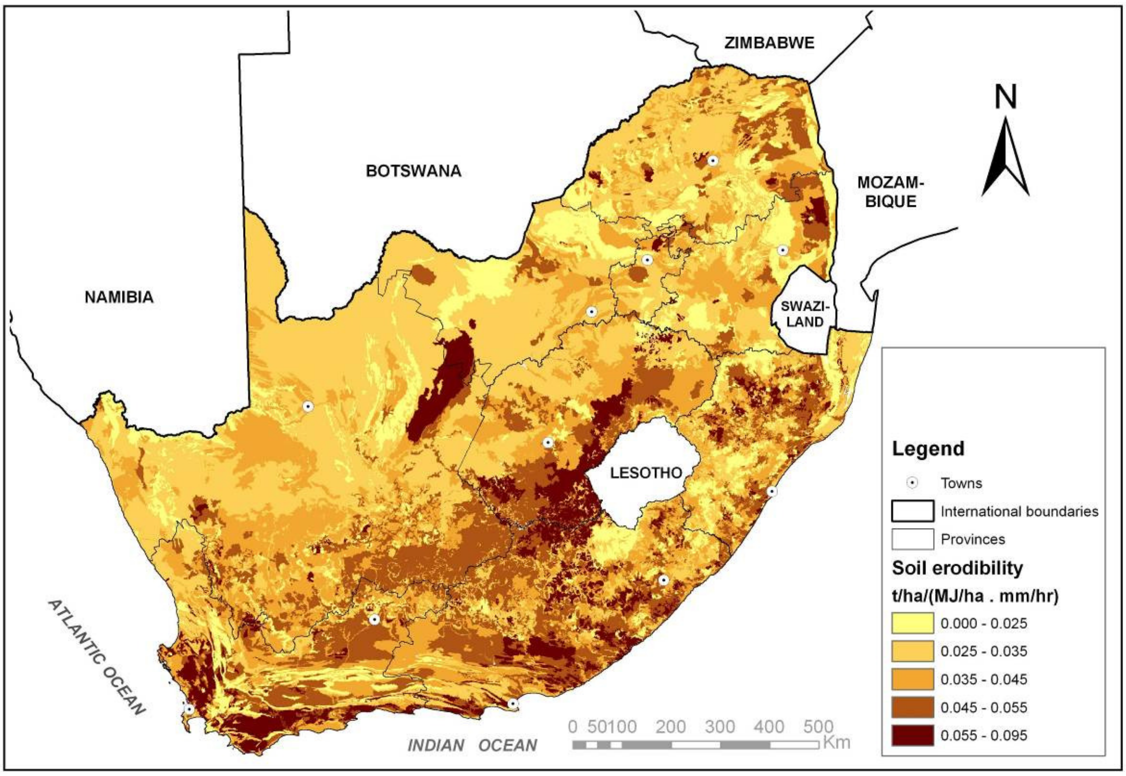

Figure 2: Soil erodibility factor (K) map of South Africa.

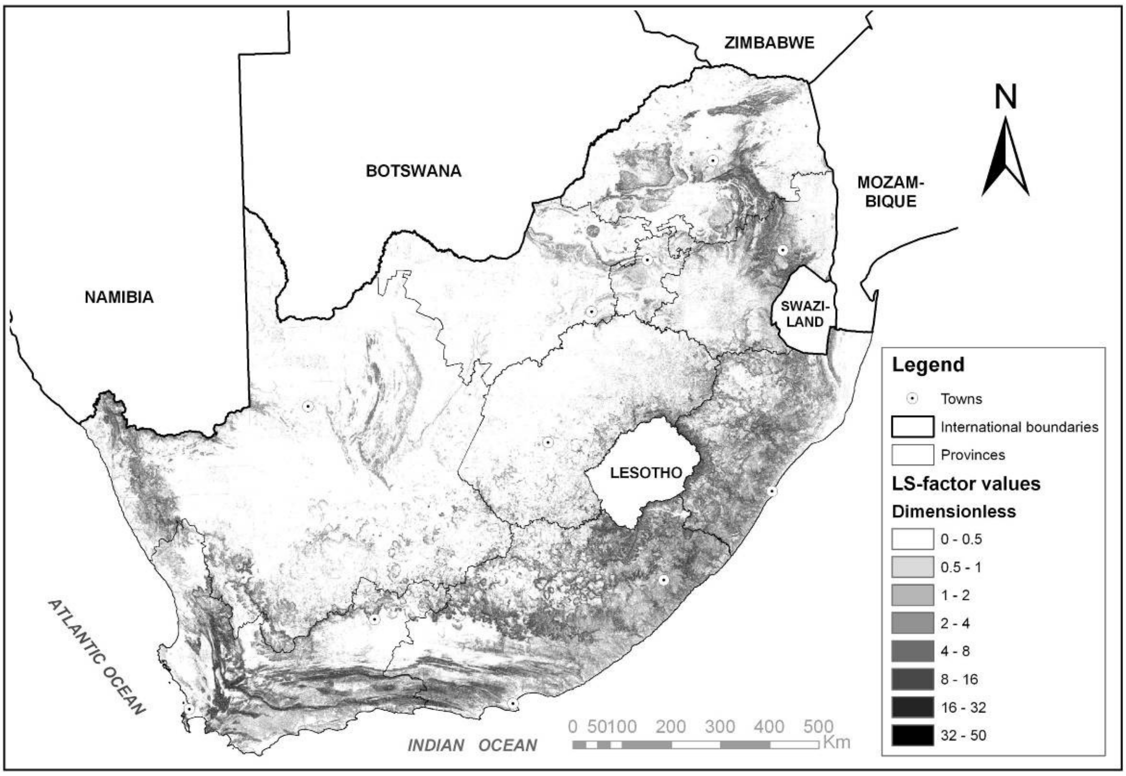

Figure 3: Topography factor (LS) map of South Africa.

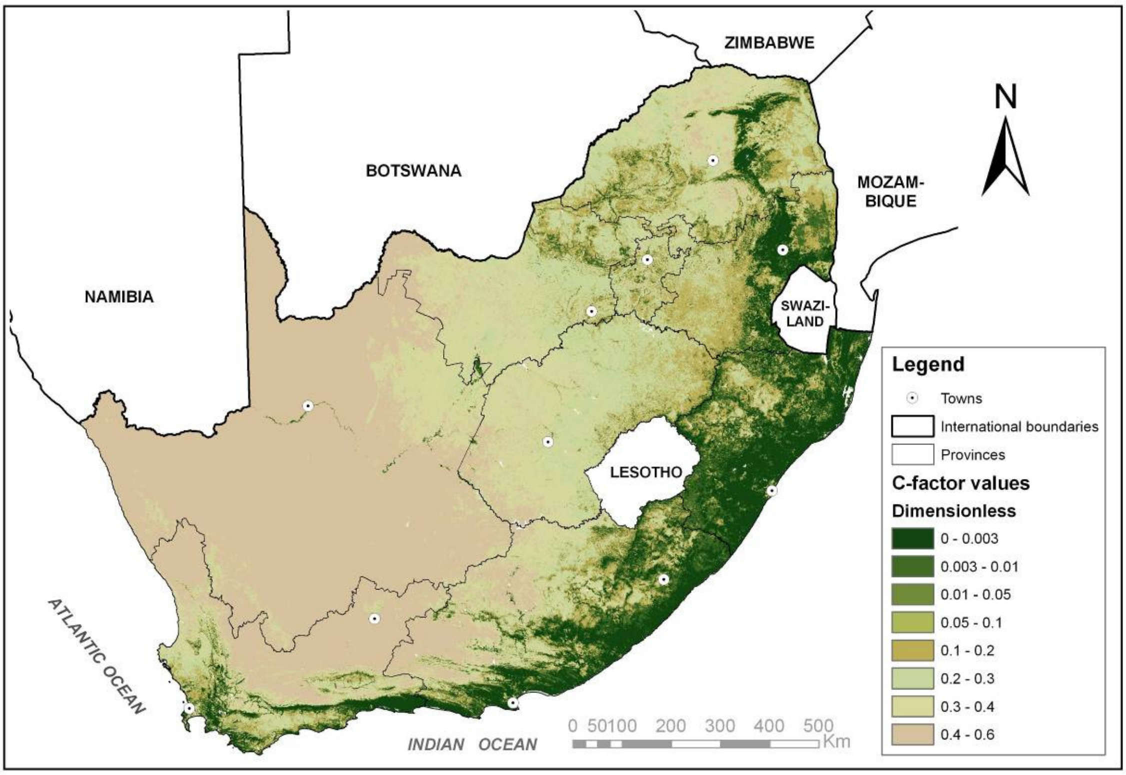

Figure 4: Cover factor map of South Africa.

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APPENDIX B:

MAPS OF FACTORS USED TO DETERMINE AREAS SUSCEPTIBLE TO GULLY EROSION

Geology, land type, soil erodibility and soil depth class layers are illustrated in Figure 5a-d respectively. The Tarkastad and Molteno Formations in the central part of the catchment are ranked as class 5, as well as the soils derived there from. Since gully development also depends on the availability of deep soils, it is not surprising that relatively large fractions of deep soils are affected by gully erosion, especially where footslopes and valleys are filled with erodible soils derived from mudstones. As mentioned in Case Study *ii* of Section 3, the soils from these Formations are associated with duplex soils that are highly erodible (class 5) with widespread gully erosion evident.

Figure 6a-e respectively illustrates slope, contributing area, the wetness and sediment transport capacity indices, as well as terrain unit class layers. Gullies are prominent on gentle footslopes in concave zones of saturation along drainage paths with large contributing areas. As mentioned above, gully formation is favoured in these areas because the critical drainage area needed for gully initiation increases as slope decreases (Poesen *et al.*, 2003), representing zones of saturation with high surface soil water along drainage paths where the contributing area is high and slope is low.

Figure 7a-b respectively illustrates vegetation cover and land use indices. Gullies are mainly located in areas with poor vegetation cover and in cultivated areas and degraded grassland. As mentioned above, gully formation is favoured in these areas because cultivated areas and degraded grassland represent areas where the soil is frequently disturbed and gully development is favoured. Field observations indicate that a relatively large portion of the cultivated and grassland areas in the catchment is affected by gully erosion due to livestock disturbance, including overgrazing and trampling along cattle tracks.

Figure 8 illustrates areas that are intrinsically susceptible to gully erosion, yet are vegetated and gully-free (estimated at approximately 7 260 ha). The identification of currently vegetated or gully-free areas susceptible to continuous and/or discontinuous gully development was also achieved (estimated at approximately 560 and 6 700 ha, respectively). Appropriate strategies need to be designed for these susceptible areas in order to protect the current vegetation cover. This approach proved to be relatively simple, realistic and practical, and it can be applied or expanded to other areas of SA at a regional scale; thereby providing a tool to help with the implementation of plans for soil conservation and sustainable management (Kheir *et al*., 2007).

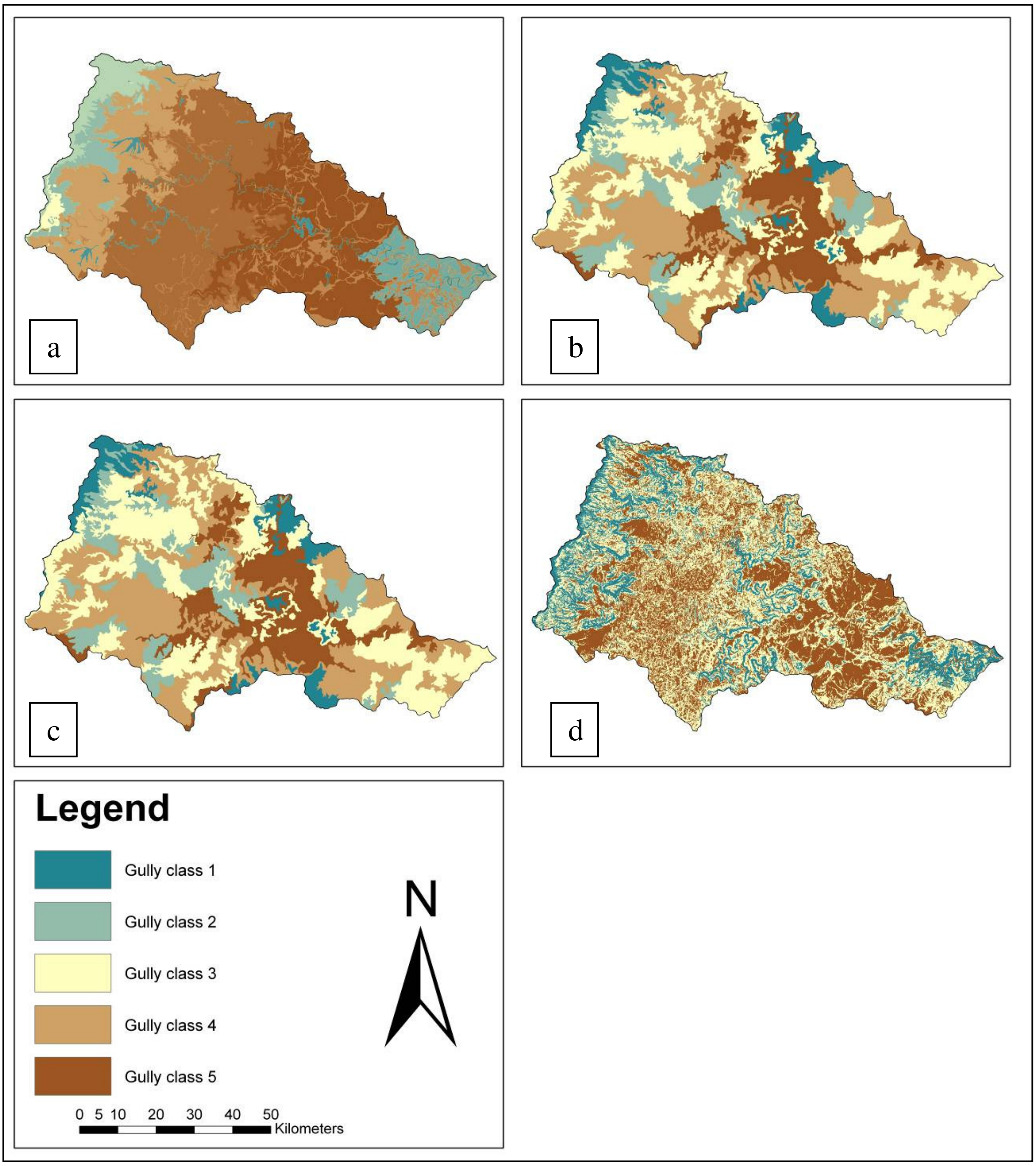

Figure 5: **Lithological and pedological gully class layers including (a) geology, (b) land type, (c) soil erodibility and (d) soil depth.**

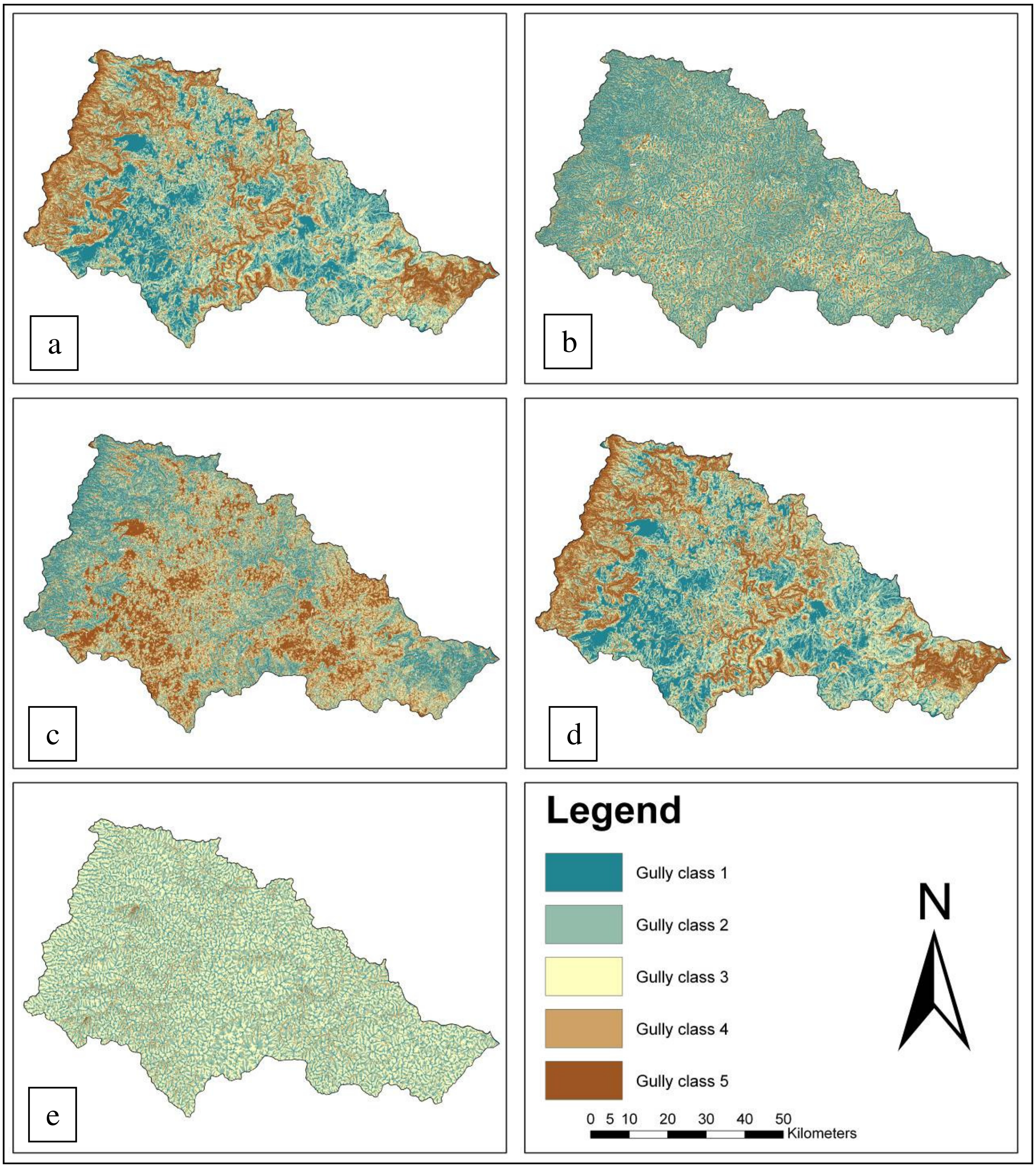

Figure 6: **Topographical gully class layers including (a) slope, (b) contributing area, (c) wetness and (d) sediment transport capacity indices, as well as (e) terrain units.**

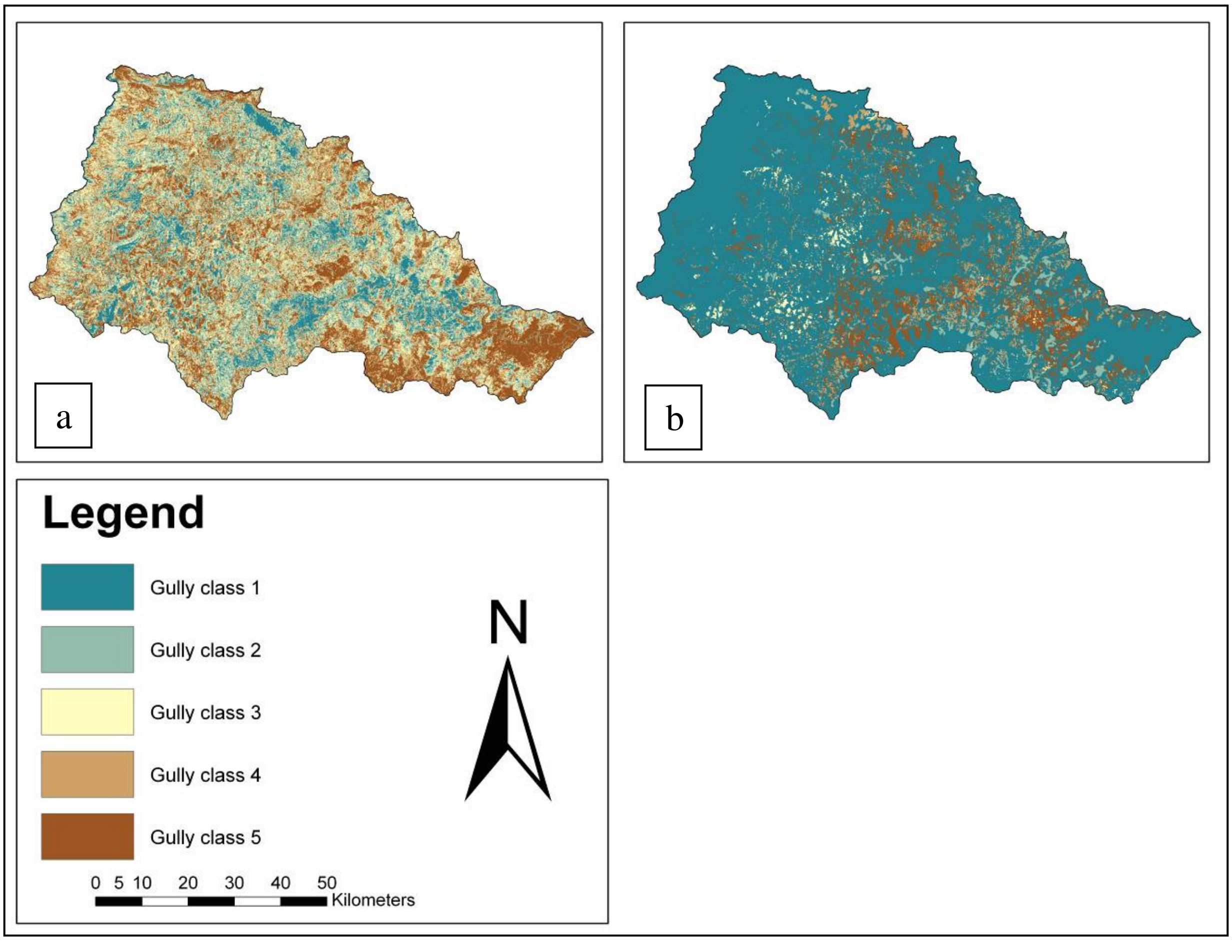

Figure 7: **(a) Vegetation and (b) land cover gully class layers.**

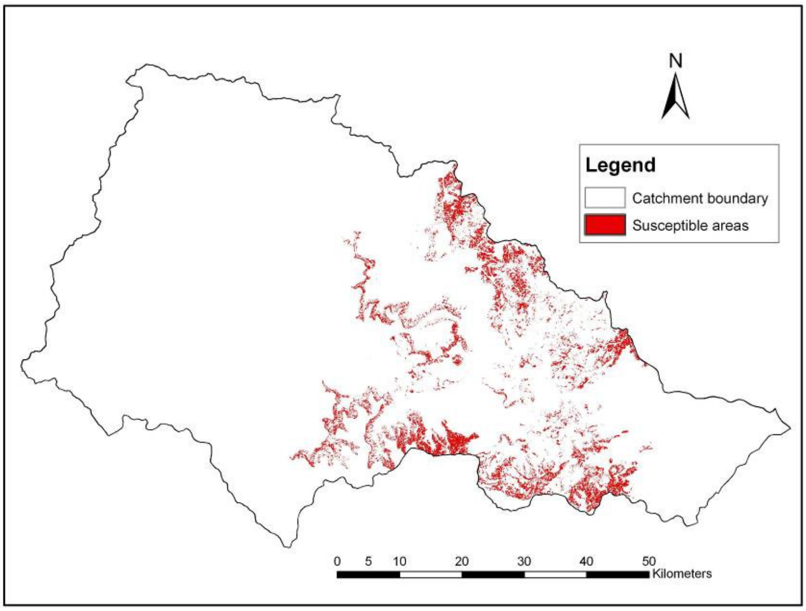

Figure 8: **Areas that are intrinsically susceptible to gully erosion, yet are vegetated and gully-free.**

References

Kheir RB, Wilson J, Deng Y. 2007. Use of terrain variables for mapping gully erosion susceptibility in Lebanon. *Earth Surface Processes and Landforms* **32**: 1770–1782.

Poesen J, Nachtergaele J, Verstraeten G, Valentin C. 2003. Gully erosion and environmental change: importance and research needs. *Catena* **50**: 91– 133.