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.

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

J.J. Le Roux
Department of Geography, Geoinformatics and Meteorology
University of Pretoria
Pretoria 0002
Current address: Institute for Soil, Climate and Water
Agricultural Research Council
Private Bag X79
Pretoria 0001

T.L. Morgenthal
Institute for Soil, Climate and Water
Agricultural Research Council
Private Bag X79
Pretoria 0001
Current address: Döhne Agricultural Development Institute
National Department of Agriculture
Private Bag X15,
Stutterheim 4930

J. Malherbe
Institute for Soil, Climate and Water
Agricultural Research Council
Private Bag X79
Pretoria 0001

D.J. Pretorius
Institute for Soil, Climate and Water
Agricultural Research Council
Private Bag X79
Pretoria 0001

P.D. Sumner
Department of Geography, Geoinformatics and Meteorology
University of Pretoria
Pretoria 0002

Published in: *Water SA*, 2008, volume 34(3), 305-314.

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.

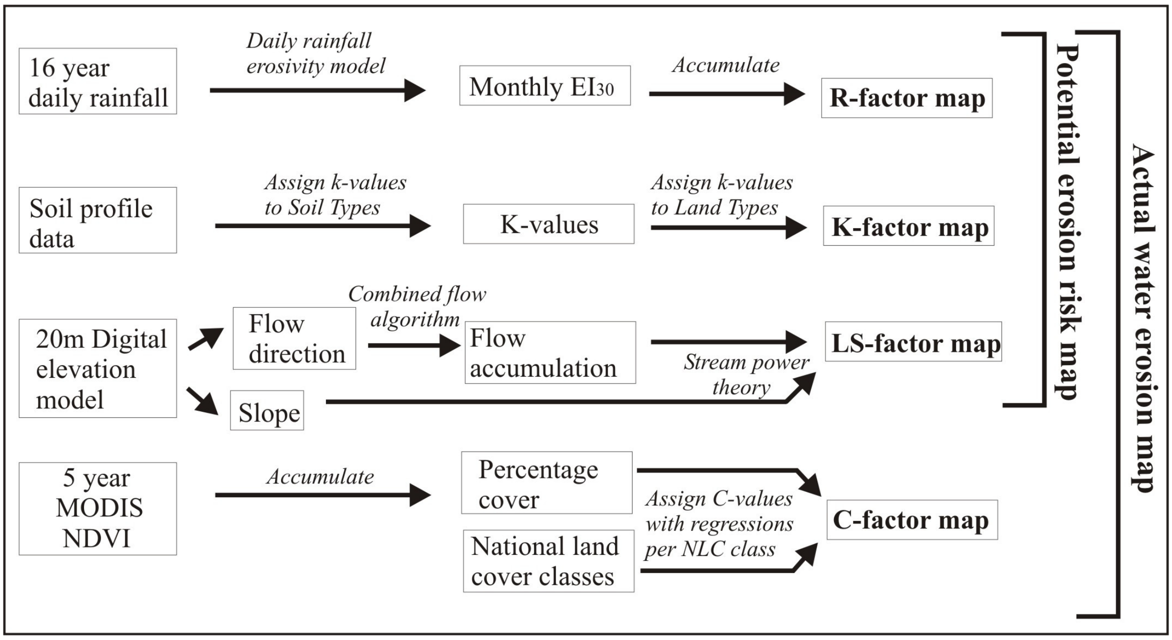

Figure 1: Methodology flow chart for mapping potential and actual water erosion.

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.

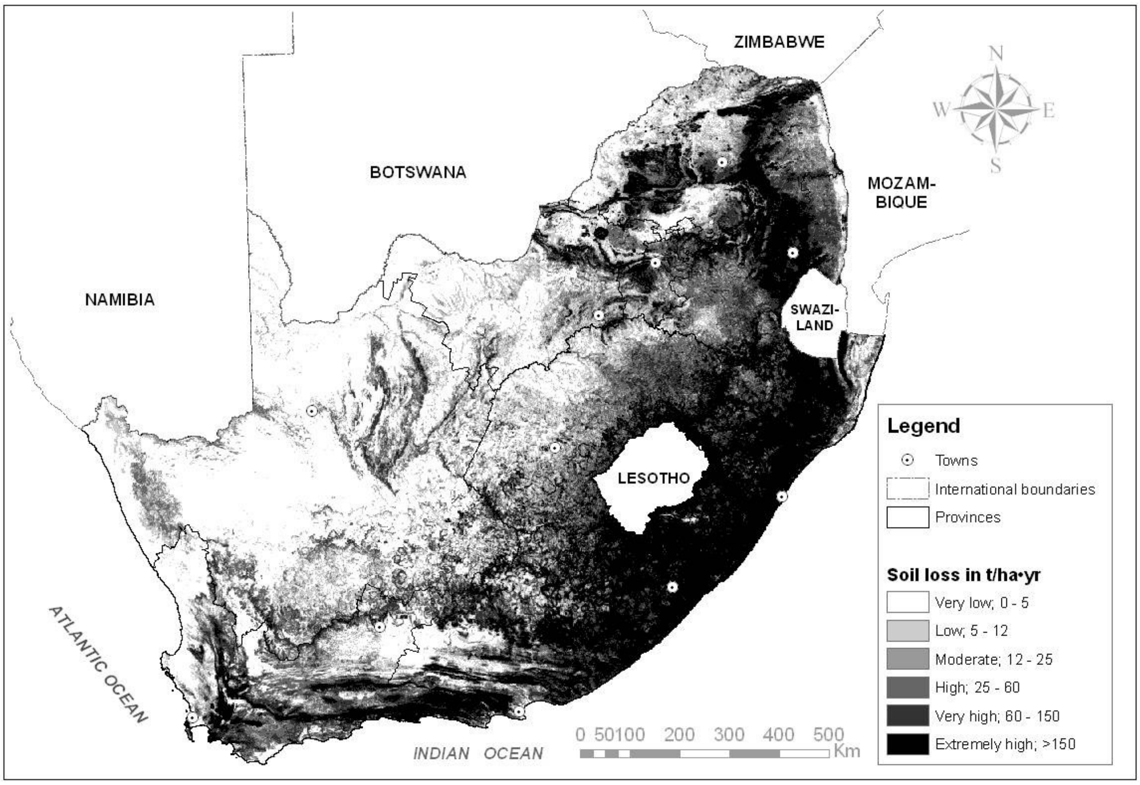

Figure 2: Potential water erosion risk map of South Africa.

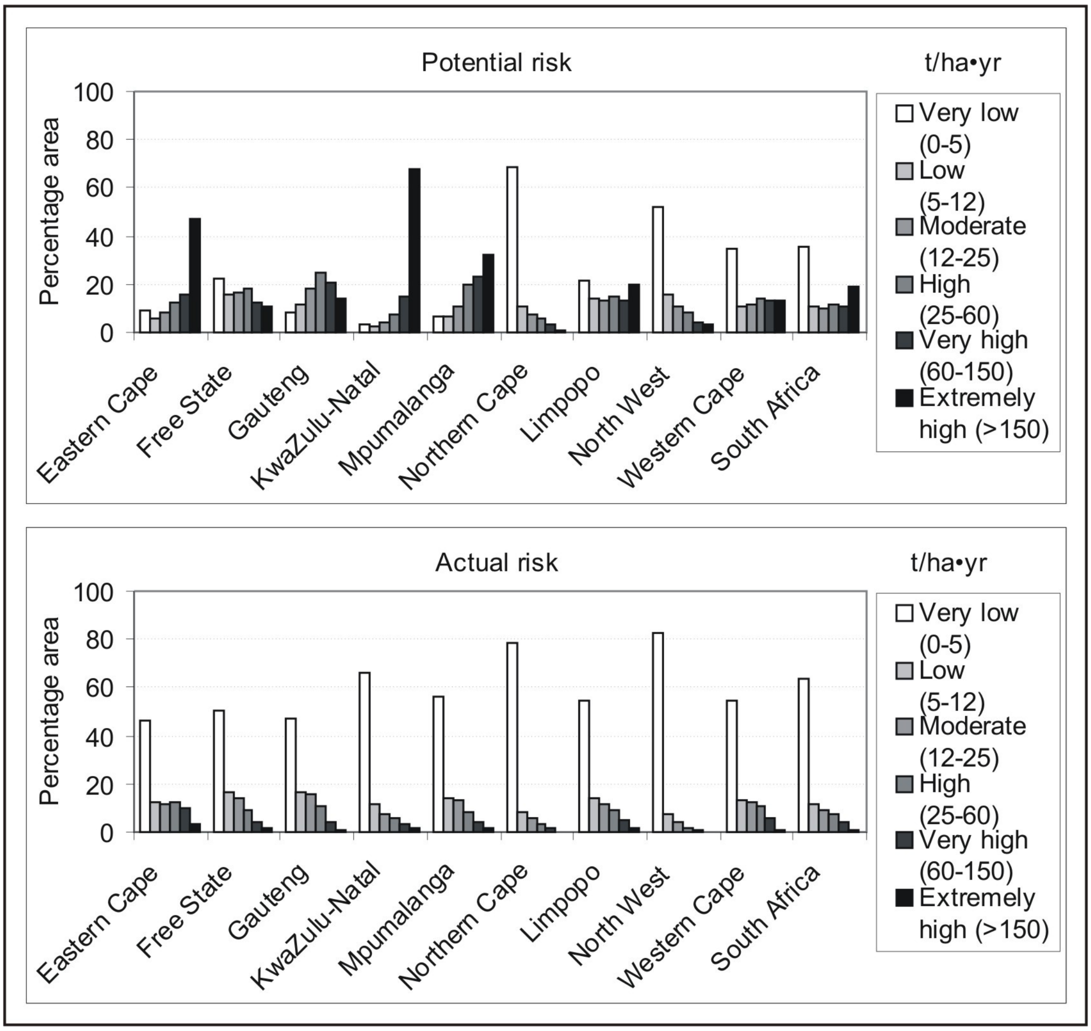

Figure 3: **Potential and actual erosion risk of each province expressed as a percentage.**

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.

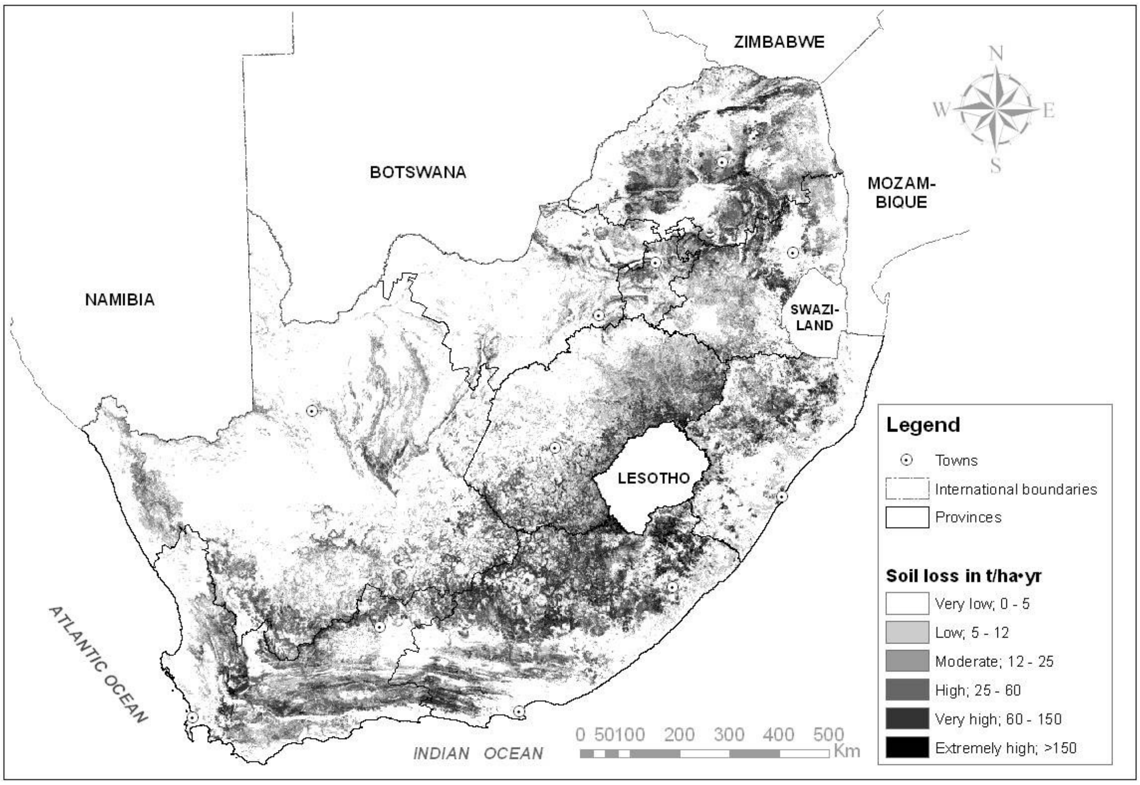

Figure 4: Actual water erosion risk map of South Africa.

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.

Acknowledgements

The authors would like to thank Mr AJ van Zyl (formerly at the ARC-ISCW) for initiating the project. The Department of Agriculture, Directorate Land Use and Soil Management (Mr HJ Lindemann) supported and funded the research for which we are thankful. The authors would like to thank Mrs M van der Walt and Mrs E van den Berg at the ARC-ISCW for providing the data required for application in the study. Thanks also to Dr HJ Smith, Mr TS Newby, Mr JL Schoeman and Mr HL Weepener of the ARC-ISCW who gave valuable advice.

The help and constant encouragement given by Mr CE Steyn is gratefully acknowledged. Editing by Dr TP Fyfield is very much appreciated.

References

Acocks JPH. 1988. Veld types of South Africa*. Memoirs of the Botanical Survey of South Africa* **57**: 1-146. Government Printer: Pretoria, South Africa.

Agrometeorology Staff. 1984-2000. *ARC-ISCW Agrometeorology Weather Station Network Data for South Africa*, *Unpublished*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Band LE, Moore ID. 1995. Scale: Landscape attributes and geographical information systems. In: Kalma JD, Sivapalan M. (eds.) *Scale Issues in Hydrological Modeling*. John Wiley & Sons: Chichester, UK. pp. 159-180.

Bergsma E, Charman P, Gibbons F, Hurni H, Moldenhauer WC, Panichapong S. 1996. *Terminology for Soil Erosion and Conservation*. ISSS, ITC, ISRIC.

Biesemans J, Van Meirvenne M, Gabriels D. 2000. Extending the RUSLE with the Monte Carlo error propagation technique to predict long-term average off-site sediment accumulation. *Journal of Soil and Water Conservation* **55**: 35-42.

Boardman J, Parsons AJ, Holland R, Holmes PJ, Washington R. 2003. Development of badlands and gullies in the Sneeuberg, Great Karoo, South Africa. *Catena* **50**: 165-184.

Bühmann C, Beukes DJ, Turner DP. 2006. Clay mineral associations in soils of the Lusikisiki area, Eastern Cape Province, and their agricultural significance. *South African Journal of Plant and Soil* **23**(2): 78-86.

Crosby CT, Smithen AA, McPhee PJ. 1983. Introduction of the Universal Soil Loss Equation in the Republic of South Africa, Paper No. 83.2072. *Proceedings of the 1983 summer meeting of the* ASAE, *June 26-29*. Bozeman, Montana, USA.

Gobin A, Govers G, Jones R, Kirkby MJ, Kosmas C, Gentile AR. 2003. *Assessment and Reporting on Soil Erosion, Background and Workshop Report, Technical Report 94*. European Environment Agency: Copenhagen, Denmark. http://reports.eea.europa.eu/technical.

Elwell HA. 1976. *Natal Agricultural Research Bulletin No. 7, Soil Loss Estimator for Southern Africa*. Department of Agricultural Technical Services: Natal, South Africa.

Flügel W, Märker M, Moretti S, Rodolfi G, Sidrochuk A. 2003. Integrating geographical information systems, remote sensing, ground thruthing and modelling approaches for regional erosion classification of semi-arid catchments in South Africa. *Hydrological Processes* **17**: 929-942.

French AN, Schmugge TJ, Kustas WP. 2000. Discrimination of senescent vegetation using thermal emissivity contrast. *Remote Sensing of the Environment* **74**: 249-254.

Gallant JC, Wilson JP. 2000. Primary topographic attributes. In: Wilson JP, Gallant JC. (eds.) *Terrain Analysis, Principles and Applications*. John Wiley & Sons, Inc: New York, USA. pp. 51-85.

Garland GG, Hoffman MT, Todd S. 2000. Soil degradation. In: Hoffman MT, Todd S, Ntshona Z, Turner S. (eds.) *A National Review of Land Degradation in South Africa*. South African National Biodiversity Institute: Pretoria, South Africa. pp. 69-107. http://www.nbi.ac.za/landdeg.

Gibson DJD. 2006. *Land Degradation in the Limpopo Province, South Africa*. MSc Dissertation. University of the Witwatersrand: Johannesburg, South Africa.

Gibson DJD, Paterson DG, Newby TS, Laker MC, Hoffman MT. 2006. Chapter 4: Land. In: Lickindorf E, Clark R. (eds.) *South Africa Environment Outlook. A Report on the State of the Environment.* Department of Environmental Affairs and Tourism: Pretoria, South Africa. pp. 87-106.

GISCOE. 2001. *GISCOE Digital Terrain Models*, *Unpublished*. GIMS: Midrand, South Africa.

Haarhoff D, Smith HJ, Beytell JF, Schoeman JL. 1994. *The Testing of Techniques for the Determination of the Erosion Potential of Areas in South Africa. ISCW Report No. GW/A/94/6*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Hoffman MT, Ashwell A. 2001. *Nature Divided. Land Degradation in South Africa*. University of Cape Town Press: Cape Town, South Africa.

Hoffman MT, Todd S. 2000. A national review of land degradation in South Africa: The influence of biophysical and socio-economic factors. *Journal of South African Studies* **26**(4): 743-758.

Hudson CA. 1987. *A Regional Application of the SLEMSA in the Cathedral Peak Area of the Drakensberg.* MSc Dissertation. University of Cape Town: Cape Town, South Africa.

Jetten V, Govers G, Hessel R. 2003. Erosion models: quality of spatial patterns. *Hydrological Processes* **17**: 887-900.

Laker MC. 2004. Advances in soil erosion, soil conservation, land suitability evaluation and land use planning research in South Africa. *South African Journal of Plant and Soil* **21**: 345-368.

Land Type Survey Staff. 1972 – 2006. *Land Types of South Africa: Digital Map (1:250 000 scale) and Soil Inventory Databases*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Le Roux JJ, Morgenthal TL, Malherbe J, Smith HJ, Weepener HL, Newby TS. 2006. *Improving Spatial Soil Erosion Indicators in South Africa. ISCW Report No. GW/A/2006/51*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Low AB, Rebelo AG. 1998. *Vegetation of South Africa, Lesotho and Swaziland*. Department of Environmental Affairs and Tourism: Pretoria, South Africa.

Lu H, Prosser IP, Moran CJ, Gallant JC, Priestley G, Stevenson JG. 2003. Predicting sheetwash and rill erosion over the Australian continent. *Australian Journal of Soil Research* **41**: 1037-1062.

MacVicar CN, De Villiers JM, Loxton RF, Lambrechts JJN, Le Roux J, Merryweather FR, Van Rooyen TH, Von M Harmse HJ, Verster E. 1977. *Soil Classification: A Binomial System for South Africa*. Department of Agricultural Technical Services: Pretoria, South Africa.

McPhee PJ, Smithen AA. 1984. Application of the USLE in the Republic of South Africa. *Agricultural Engineering in South Africa* **18**: 5-13.

Merritt WS, Letcher RA, Jakeman AJ. 2003. A review of erosion and sediment transport models. *Environmental Modelling and Software* **18**: 761-799.

Moore ID, Burch GJ. 1986. Physical basis of the length-slope factor in the Universal Soil Loss Equation (USLE). *Soil Science Society of America Journal* **50**: 1294-1298.

Moore ID, Wilson JP. 1992. Length-slope factors for the Revised Universal Soil Loss Equation (RUSLE) simplified method of estimation. *Journal of Environmental Quality* **22**: 458-466.

Morgan RPC. 1995. *Soil Erosion and Conservation (2nd edn.)*. Longman Group Limited: London, UK.

Morgenthal TL, Malherbe J, Van Zyl D, Newby TS. 2006. *Evaluation of MODIS Satellite Data as an Assessment Tool to Audit the State of Agricultural Resources. ISCW Report No. GW/A/2006/72*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

National Land Cover. 2000. *National Land Cover 2000*, *Unpublished*. ARC-Institute for Soil, Climate and Water; Council for Scientific and Industrial research: Pretoria, South Africa.

Nearing MA. 1997. A single continuous function for slope steepness influence on soil loss. *Soil Science Society of America Journal* **61**: 917-919.

NRI. 2001. *National Resources Inventory, 2001 Annual NRI, Soil Erosion*. United States Department of Agriculture – National Resources Conservation Service: USA. http://www.nrcs.usda.gov.

Oldeman LR, Hakkeling RTA, Sombroek WG. 1991. *GLASOD World Map of the Status of Human-Induced Soil Degradation (2nd edn.)*. ISRIC: Wageningen. UNEP: Nairobi.

Pretorius DJ. 1995. *The Development of a Soil Degradation Management Support System, Unpublished ISCW Report*. ARC- Institute for Soil, Climate and Water: Pretoria, South Africa.

Pretorius DJ. 1998. The Development of Land Degradation Monitoring and Auditing *Techniques with the Aid of Remote Sensing and GIS Technology. ISCW Report No. GW/A/98/27*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Pretorius DJ, Bezuidenhout CJ. 1994. *Report on the Development of a Methodology to Determine the Nature, Rate and Extent of Soil Erosion in South Africa. ISCW Report No. GW1A/94/7*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Pretorius DJ, Smith HJ. 1998. The development of a GIS based land information system for developing areas. *Proceedings of the 21st SSSSA Conference, 21-23 January*: Drakensberg, South Africa.

Prosser IP, Rutherford ID, Olley JM, Young WJ, Wallbrink PJ, Moran CJ. 2001. Large-scale patterns of erosion and sediment transport in river networks, with examples from Australia. *Marine and Freshwater Research* **52**: 81-99.

Renard KG, Foster GR, Weesies GA, McCool DK, Yoder DC. 1994. *RUSLE Users Guide. Predicting Soil Erosion by Water: A Guide to Conservation Planning with the Revised Universal Soil Loss Equation. USDA, Agriculture Handbook No. 703*. USDA: Washington DC, USA.

Rooseboom A, Verster E, Zietsman HL, Lotriet HH. 1992. *The Development of the New Sediment Yield Map of South Africa. WRC Report No. 297/2/92*. Water Research Commission: Pretoria, South Africa.

Schäuble H. 2003. *Hydrotools 1.0 for ArcView 3.x*. Technical University of Darmstadt: Darmstadt, Germany. http://arcscripts.esri.com.

Schulze RE. 1979. Soil loss in the key area of the Drakensberg – A regional application of the Soil Loss Estimation Model for Southern Africa (SLEMSA). In: *Hydrology and Water Resources of the Drakensberg*. Natal and Regional Planning Commission: Pietermaritzburg, South Africa. pp. 149-167.

Scott DF, Schulze R. 1991. The hydrological effects of a wildfire in a eucalypt afforested catchment. *South African Forestry Journal* **160**: 67-74.

Smith HJ. 1999. Application of empirical soil loss models in southern Africa: A review. *South African Journal of Plant and Soil* **16**(3): 158-163.

Smith HJ, Haarhof D, Schoeman JL. 1995. *Die Kwantitatiewe Beraming van Bronverlies Onder Verskillende Toestande in die Sekhukhune-Distrik in Lebowa. ISCW Report No. GW/W/96/17*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Smith HJ, Van Zyl AJ, Claasens AS, Schoeman JL, Laker MC. 2000. Soil loss modelling in the Lesotho Highlands Water Project catchment areas. *South African Geographical Journal* **82**: 64-69.

Smithen AA. 1981. *Characteristics of Rainfall Erosivity in South Africa.* MSc Dissertation. University of Natal: Pietermartizburg, South Africa.

Smithen AA, Schulze RE. 1982. The spatial distribution in Southern Africa of Rainfall Erosivity for use in the Universal Soil Loss Equation. *Water SA* **8**(2): 74-78.

Snyman HA, Van Rensburg WLJ, Opperman DPJ. 1986. Toepassing van 'n gronderosievergelyking op natuurlike veld van die Sentrale Oranje-Vrystaat. *Journal of the Grassland Society of southern Africa*. **3**: 4-9.

Snyman HA. 1999. Soil erosion and conservation. In: Tainton NM (ed.) *Veld Management in South Africa*. University of Natal Press: Scottsville, South Africa. pp. 355-380.

Soil Survey Staff. 1973 – 1987. *Soil Profile Descriptions and Soil Analyses Data. ARC-ISCW Soil Profile Information System*, *Unpublished*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Ströhmenger PHE, Van Der Merwe JPA, Smith HJ, Van Den Berg HM, Van Den Berg EC, Dekker C, Malherbe J, De Nysschen G, Van Der Walt M, Haasbroek P, Morgenthal TL, Kellner K, Van Rensburg L. 2004. *Auditing the Conservation Status of the Natural Resources in the OR Tambo and Umkhanyakude ISRDS Nodes. ISCW Report No. GW/A/2003/47/2*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Vrieling A. 2006. Satellite remote sensing for water erosion assessment: A review. *Catena* **65**: 2-18.

Wessels KJ, Van Den Berg HM, Van Der Merwe JPA, Smith HJ, Van Zyl AJ, Twyman L. 2001a. *Mapping and Monitoring the Conservation Status of the Natural Resources of Mpumalanga Province by Means of Remote Sensing and GIS Technology. ISCW Report No. GW/A/2001/9*, ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Wessels KJ, Van Der Merwe JPA, Smith HJ, Ströhmenger PHE, Van Zyl AJ, Twyman L. 2001b. *Gauteng Natural Resources Audit. ISCW Report No. GW/A/2001/41*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Wischmeier WH, Smith DD. 1978. *Predicting Rainfall Erosion Losses, a Guide to Conservation Planning. USDA Agriculture Handbook No. 537*. USDA: Washington DC, USA.

Yu B, Rosewell CJ. 1996a. An assessment of a daily rainfall erosivity model for New South Wales. *Australian Journal of Soil Research* **34**: 139-152.

Yu B, Rosewell CJ. 1996b. Rainfall erosivity estimations using daily rainfall amounts for South Australia. *Australian Journal of Soil Research* **34**: 721-733.

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

J.J. Le Roux
Department of Geography, Geoinformatics and Meteorology
University of Pretoria
Pretoria 0002
Current address: Institute for Soil, Climate and Water
Agricultural Research Council
Private Bag X79
Pretoria 0001

P.D. Sumner
Department of Geography, Geoinformatics and Meteorology
University of Pretoria
Pretoria 0002

In press: *Land Degradation and Development*, 2011, DOI: 10.1002/ldr.1083.

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).

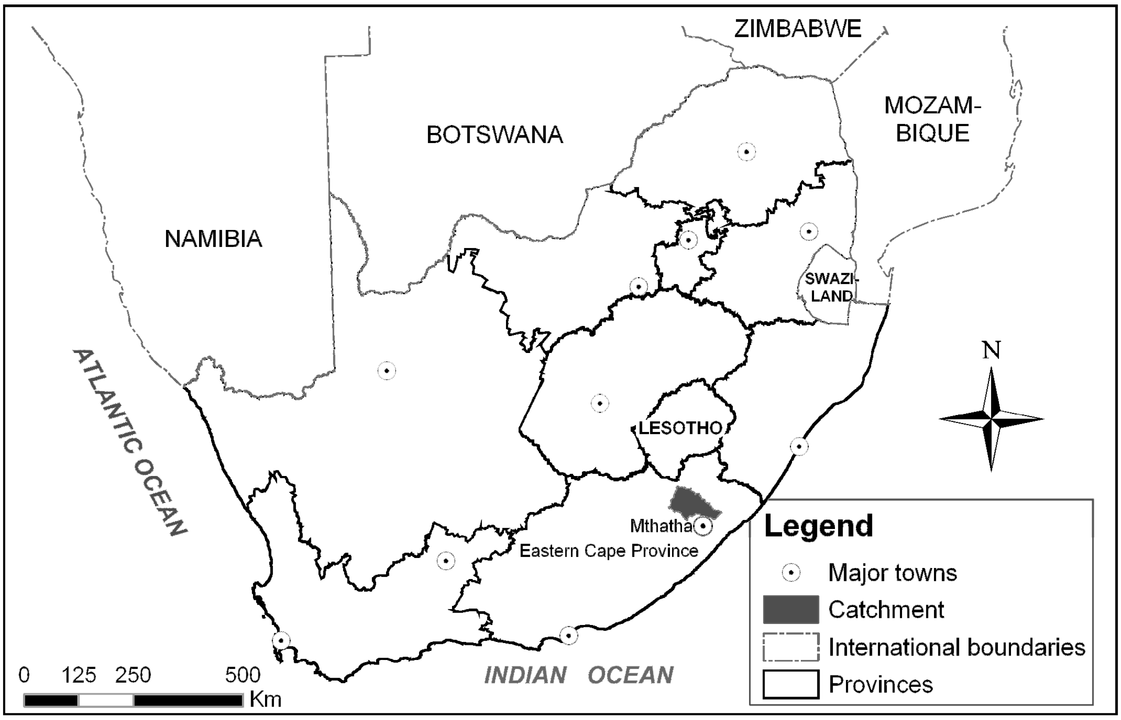

Figure 1: **Location map of study area in the Eastern Cape Province, South Africa.**

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).

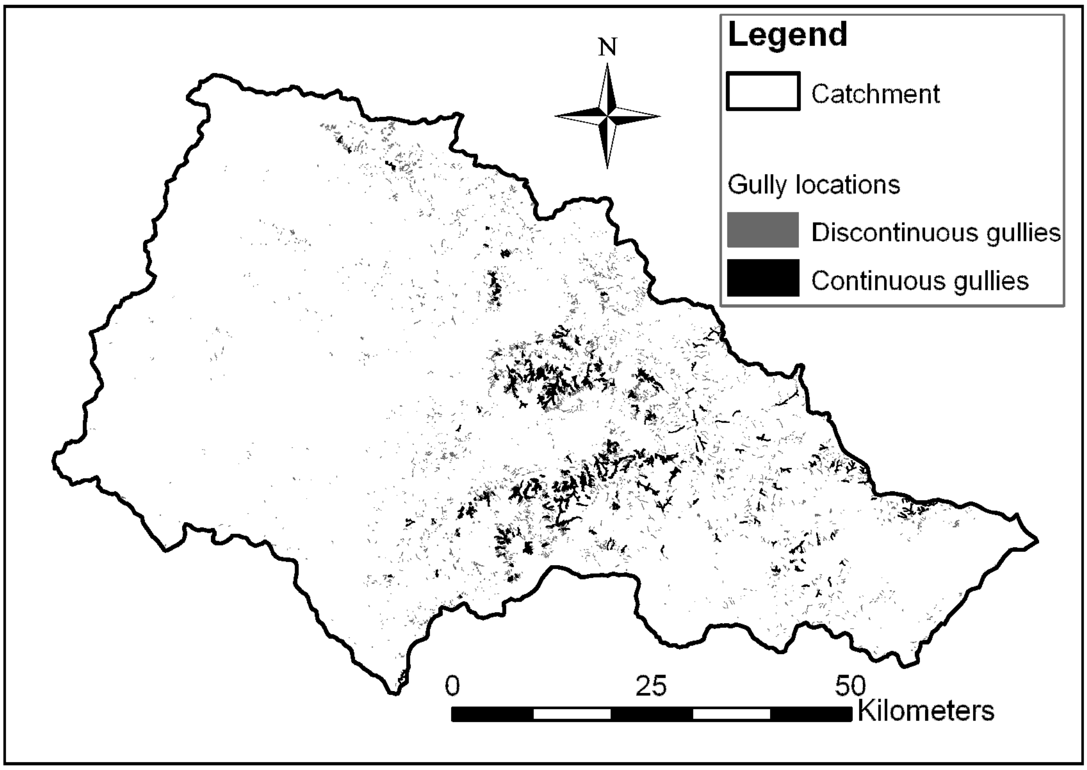

Figure 2: **Gully locations map of the catchment in the Eastern Cape Province, South Africa.**

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.

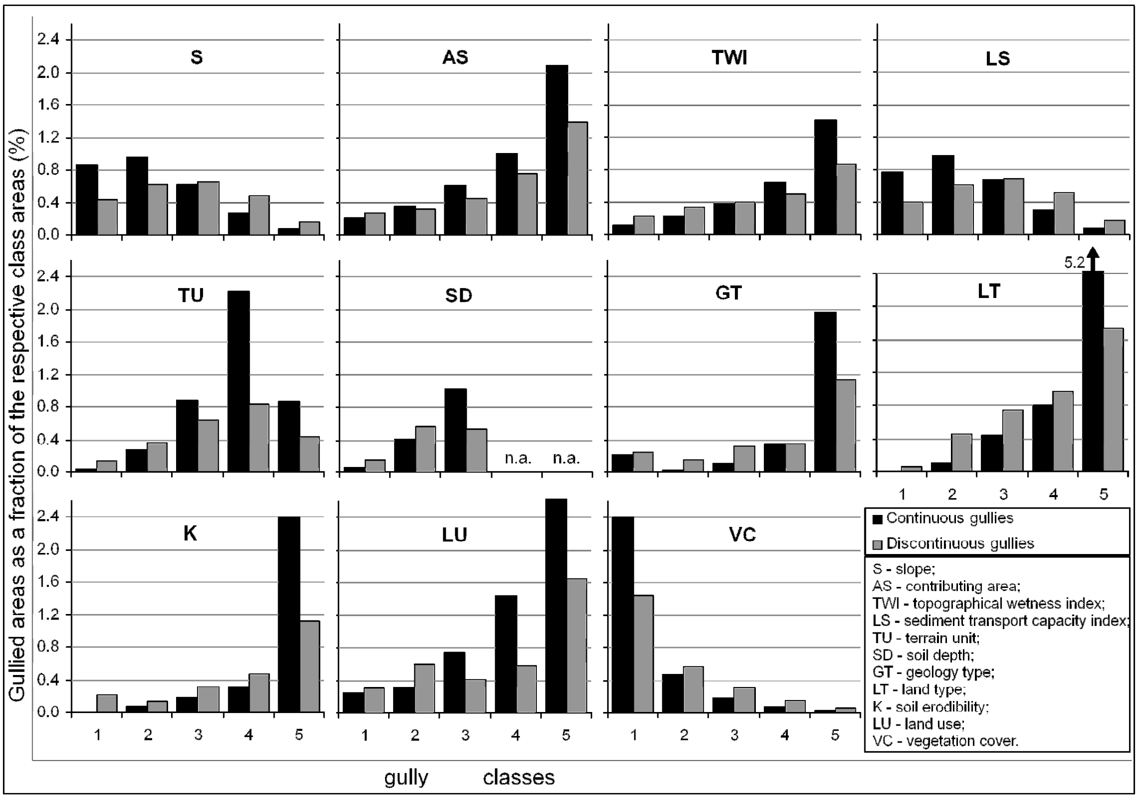

Figure 3: Continuous and discontinuous gullied areas of each class (1-5) as a fraction of the respective class area.

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.

Acknowledgements

Numerous colleagues at the Agricultural Research Council - Institute for Soil, Climate and Water (ARC-ISCW) have assisted the research efforts including Mr. H.L. Weepener, Mrs. K. Middleton, Mr. J.L. Schoeman, Dr. H.J. Smith and Mr. J.C.L. Potgieter, as well as literature provided by Mrs. R. van Dyk. Special thanks to Mr. Z.E. Mashimbye for digitizing some of the gullies utilized in the study. Thanks to Mr. H.M. Van Den Berg of IRIS International for valuable advice. The authors are thankful for research funding from the ARC-ISCW, as well as Mr. D.J. Pretorius and his colleagues at the Department of Agriculture Forestry and Fisheries (DAFF), Directorate Land Use and Soil Management. The paper benefited greatly from the comments by two referees on an earlier version.

References

Beckedahl HR. 1998. Aspects of subsurface erosion phenomena in eastern southern Africa. *Peterman's Geographische Mitteilungen (Supplement)* **245**: 213.

Beckedahl HR, Dardis GF. 1988. The role of artificial drainage in the development of soil pipes and gullies, some examples from Transkei, southern Africa. In: Dardis GF, Moon BP. (eds.) *Geomorphological studies in southern Africa*. Balkema: Rotterdam. pp. 229-245.

Boardman J, Foster I. 2008. Badland and gully erosion in the Karoo, South Africa. *Journal of Soil and Water Conservation* **63**(4): 121A-125A.

Council for Geoscience. 2007. *Geological data 1:250 000*. Council for Geoscience: Pretoria, South Africa.

Descroix L, González Barrios JL, Viramontes D, Poulenard J, Anaya E, Esteves M, Estrada J. 2008. Gully and sheet erosion on subtropical mountain slopes: Their respective roles and the scale effect. *Catena* **72**: 325–339.

Desmet PJJ, Poesen J, Govers G, Vandaele K. 1999. Importance of slope gradient and contributing area for optimal prediction of the initiation and trajectory of ephemeral gullies. *Catena* **37**: 377–392.

Elwell HA. 1976. *Natal Agricultural Research Bulletin No 7, Soil Loss Estimator for Southern Africa*. Department of Agricultural Technical Services: Natal, South Africa.

Evans IS. 1979. *An integrated system of terrain analysis and slope mapping*. Final report on grant DA-ERO-591-73-G0040. University of Durham: England. http://www.soi.city.ac.uk/~jwo/phd.

Flügel W, Märker M, Moretti S, Rodolfi G, Sidrochuk A. 2003. Integrating geographical information systems, remote sensing, ground truthing and modeling approaches for regional erosion classification of semi-arid catchments in South Africa. *Hydrological Processes* **17**: 929-942.

GISCOE. 2001. *GISCOE Digital Terrain Models*, *Unpublished*. GIMS: Midrand, South Africa.

Gutiérrez AG, Schnabel S, Felicísimo AM. 2009. Modelling the occurrence of gullies in rangelands of southwest Spain. *Earth Surface Processes and Landforms* **34**: 1894–1902.

Kakembo V, Xanga WW, Rowntree K. 2009. Topographic thresholds in gully development on the hillslopes of communal areas in Ngqushwa Local Municipality, Eastern Cape, South Africa. *Geomorphology* **110**: 188–194.

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.

Kheir RB, Chorowicz J, Abdallah C, Dhont D. 2008. Soil and bedrock distribution estimated from gully form and frequency: A GIS-based decision-tree model for Lebanon. *Geomorphology* **93**: 482–492.

Killick DJB. 1963. *An account of the plant ecology of the Cathedral Peak area of the Natal Drakensberg, Memoirs of the Botanical Survey of South Africa, 32*. Department of Agricultural Technical Service: Pretoria, South Africa.

Kirkby MJ, Bracken LJ. 2009. Gully processes and gully dynamics. *Earth Surface Processes and Landforms* **34**: 1841–1851.

Laker MC. 2004. Advances in soil erosion, soil conservation, land suitability evaluation and land use planning research in South Africa. *South African Journal of Plant and Soil* **21**: 345-368.

Land Type Survey Staff. 1972–2008. *Land Types of South Africa: Digital Map (1:250 000 scale) and Soil Inventory Databases*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Le Roux JJ, Mashimbye ZE, Weepener HL, Newby TS, Pretorius DJ. 2008a. *Erosion status of priority tertiary catchment areas identified by the Soil Protection Strategy of the Department of Agriculture. ISCW Report No. GW/A/2008/17*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

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

Low AB, Rebelo AG. 1998. *Vegetation of South Africa, Lesotho and Swaziland*. Department of Environmental Affairs and Tourism: Pretoria, South Africa.

Martinez-Casasnovas JA. 2003. A spatial information technology approach for the mapping and quantification of gully erosion. *Catena* **50**: 293-308.

Martinez-Casasnovas JA, Anton-Fernandez C, Ramos MC. 2003. Sediment production in large gullies of the Mediterranean area (NE Spain) from high-resolution digital elevation models and geographical information systems analysis. *Earth Surface Processes and Landforms* **28**: 443–456.

Mitasova H, Zlocha M, Hofierka J, Iverson LR. 1996. Modelling topographic potential for erosion and deposition using GIS. *International Journal of Geographical Information Systems* **10**(5): 629-641.

National Land Cover. 2000. *National Land Cover 2000*, *Unpublished*. ARC-Institute for Soil, Climate and Water; Council for Scientific and Industrial research: Pretoria, South Africa.

Ndomba PM, Mtalo F, Killingtveit A. 2009. Estimating gully erosion contribution to large catchment sediment yield rate in Tanzania. *Physics and Chemistry of the Earth* **34**: 741–748.

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

Rienks SM, Botha GA, Hughes JC. 2000. Some physical and chemical properties of sediments exposed in a gully (donga) in northern KwaZulu-Natal, South Africa and their relationship to the erodibility of the colluvial layers. *Catena* **39**: 11–31.

Rodriguez E, Morris CS, Belz JE, Chapin EC, Martin JM, Daffer W, Hensley S. 2005. *An assessment of the SRTM topographic products, Technical Report JPL D-31639*. Jet Propulsion Laboratory: Pasadena, USA.

Samadi M, Germishuyse T, Van der Walt M. 2005. *Understanding South African Soils*. ARC – Institute for Soil, Climate and Water: Pretoria, South Africa.

Schmidt J, Evans IS, Brinkmann J. 2003. Comparison of polynomial models for land surface curvature calculation. *International Journal of Geographical Information Science* **17**(8): 797-814.

Sonneveld MPW, Everson TM, Veldkamp A. 2005. Multi-scale analysis of soil erosion dynamics in Kwazulu-Natal, South Africa. *Land Degradation and Development* **16**: 287–301.

Tamene L, Park SJ, Dikau R, Vlek PLG. 2006. Analysis of factors determining sediment yield variability in the highlands of northern Ethiopia. *Geomorphology* **76**: 76– 91.

Tarboton DG. 2004. TAUDEM. *Terrain Analysis using digital elevation models*. Utah State University: Utah, USA.

Taruvinga K. 2008. *Gully mapping using remote sensing: Case study in KwaZulu-Natal, South Africa.* MSc Dissertation. Waterloo University: Canada.

Valentin C, Poesen J, Li Y. 2005. Gully erosion: Impacts, factors and control. *Catena* **63**: 132–153.

Van Den Berg HM, Weepener HL. 2009. *Development of spatial modelling methodologies for semi-detailed soil mapping, primarily in support of curbing soil degradation and the zoning of high potential land. ISCW Report No. GW/A/2009/01*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Vrieling A. 2006. Satellite remote sensing for water erosion assessment: A review. *Catena* **65**: 2-18.

Vrieling A, Rodrigues SC, Bartholomeus H. Sterk G. 2007. Automatic identification of erosion gullies with ASTER imagery in the Brazilian Cerrados. *International Journal of Remote Sensing* **28**(12): 2723-2738.

Watson HK, Ramokgopa R. 1997. Factors influencing the distribution of gully erosion in KwaZulu-Natal's Mfolozi catchment – land reform implications. *South African Geographical Journal* **79**(1): 27-34.

Wischmeier WH, Smith DD. 1978. *Predicting Rainfall Erosion Losses, a Guide to Conservation Planning. USDA Agriculture Handbook No. 537*. USDA: Washington DC, USA.

Wilson JP, Gallant JC. 2000. Digital terrain analysis. In: Wilson JP, Gallant JC. (eds.) *Terrain analysis*. John Wiley & Sons, Inc: New York. pp. 1-27.

Zhang X, Drake N, Wainwright J. 2002. Scaling land surface parameters for global-scale soil erosion estimation. *Water Resources Research* **38**(9): 191-199.

UNIVERSITEIT VAN PRETORIA
UNIVERSITY OF PRETORIA
YUNIBESITHI YA PRETORIA

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

J.J. Le Roux
Department of Geography, Geoinformatics and Meteorology
University of Pretoria
Pretoria South Africa 0002
Current address: Institute for Soil, Climate and Water
Agricultural Research Council
Private Bag X79
Pretoria 0001

P.D. Sumner
Department of Geography, Geoinformatics and Meteorology
University of Pretoria
Pretoria South Africa 0002

S. Lorentz
School of Bioresources Engineering and Environmental Hydrology
University of KwaZulu-Natal
Scottsville South Africa 3209

T. Germishuyse
Golder Associates Africa (Pty) Ltd
Midrand South Africa 1685

In press: *Geosciences*, 2012, volume 2(5).

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.

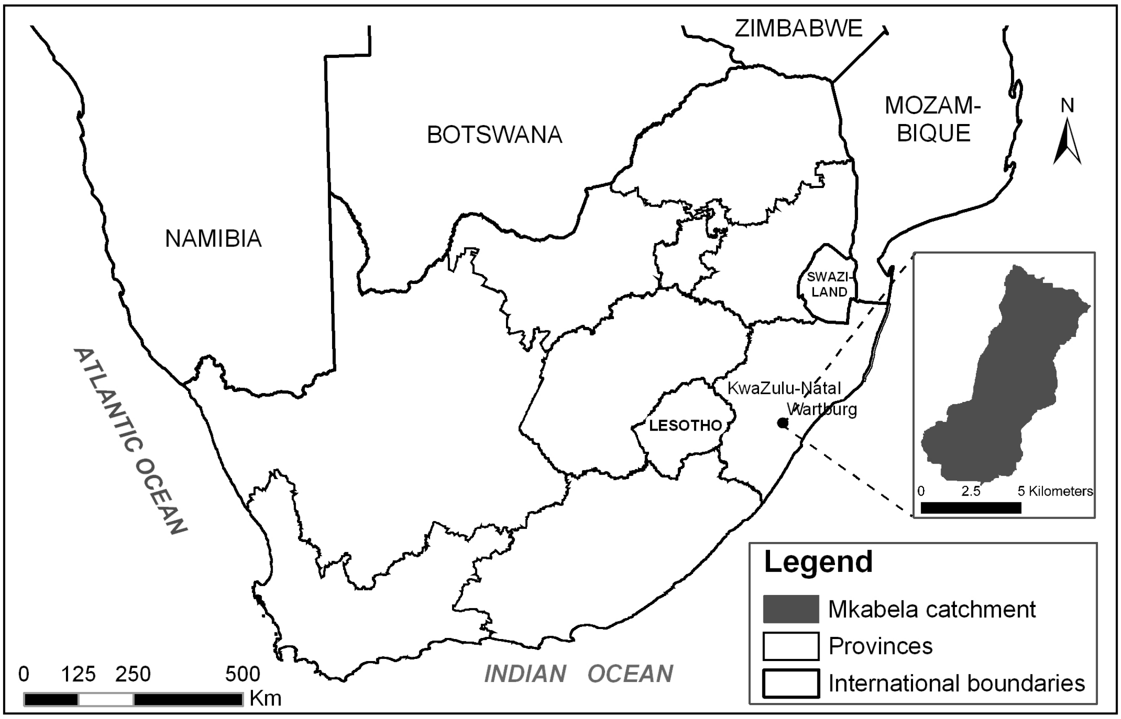

Figure 1: **Location map of Mkabela catchment in the KwaZulu-Natal Province, South Africa.**

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).

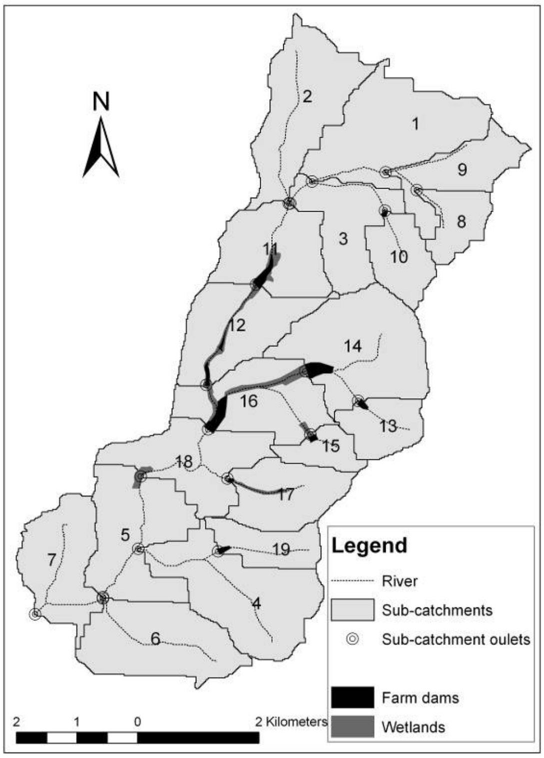

Figure 2: **Sub-catchment boundaries, outlets, location of river channel, farm dams and wetlands.**

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.

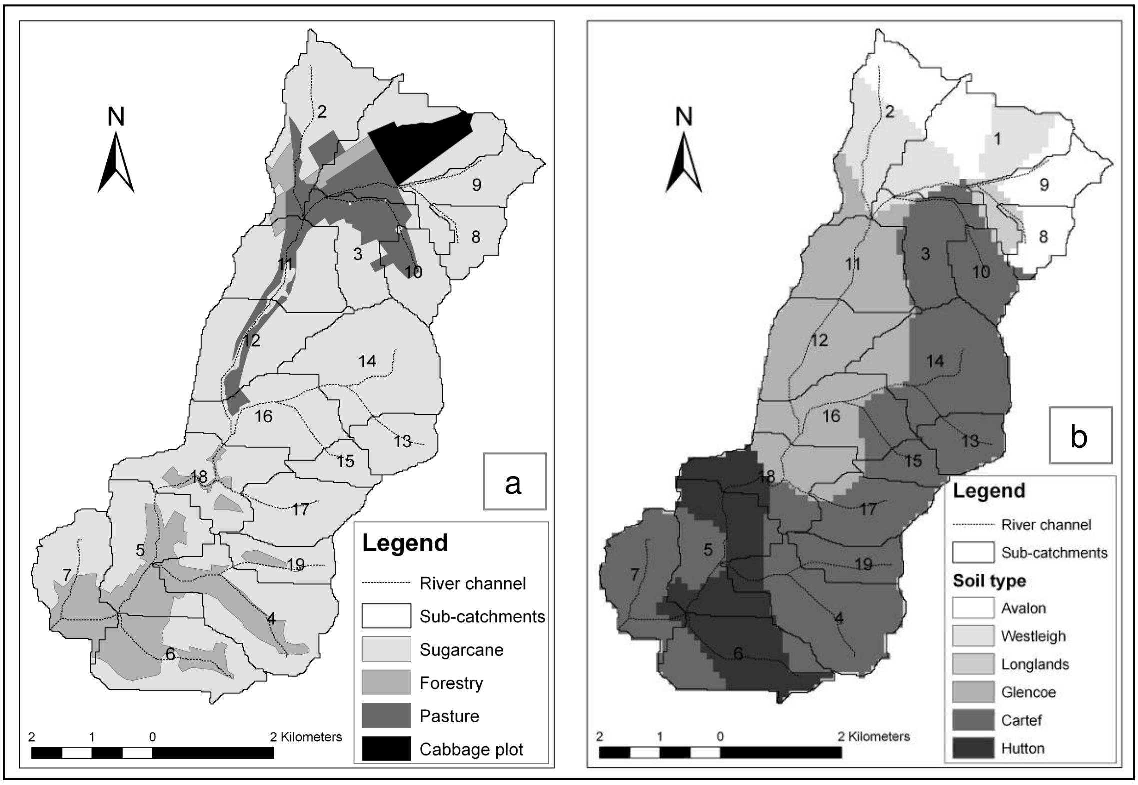

Figure 3: **(a) Land cover map and (b) soil map of Mkabela catchment (after Le Roux *et al.*, 2006).**

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.

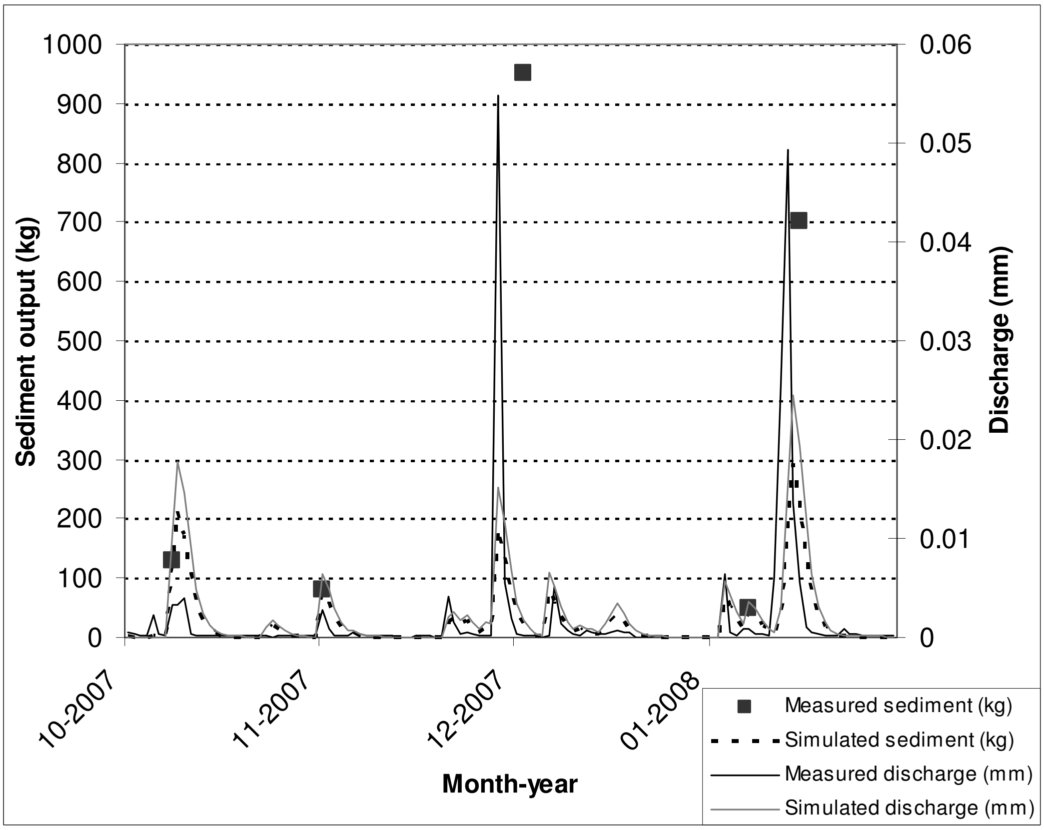

Figure 4: **Observed and simulated discharge and sediment loads of 5 rainfall events that occurred from October 2007 to January 2008.**

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.

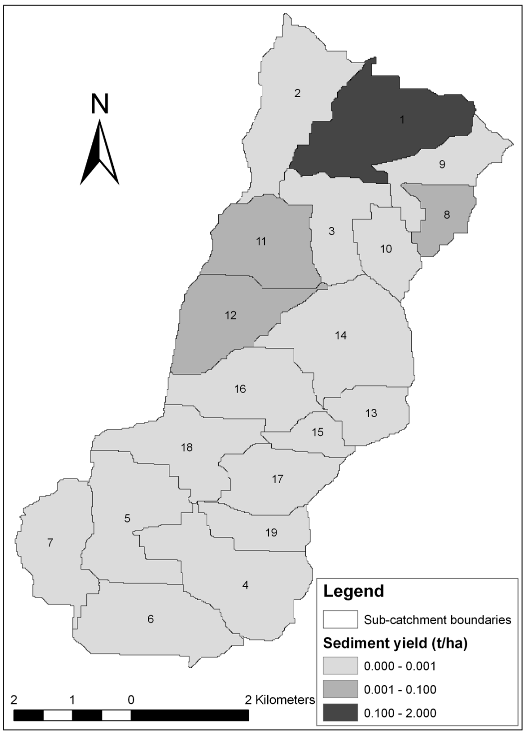

Figure 5: **Sediment yield per sub-catchment (in t/ha) that is transported into the channel during the observation period (1 July 2006 to 30 June 2008).**

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.

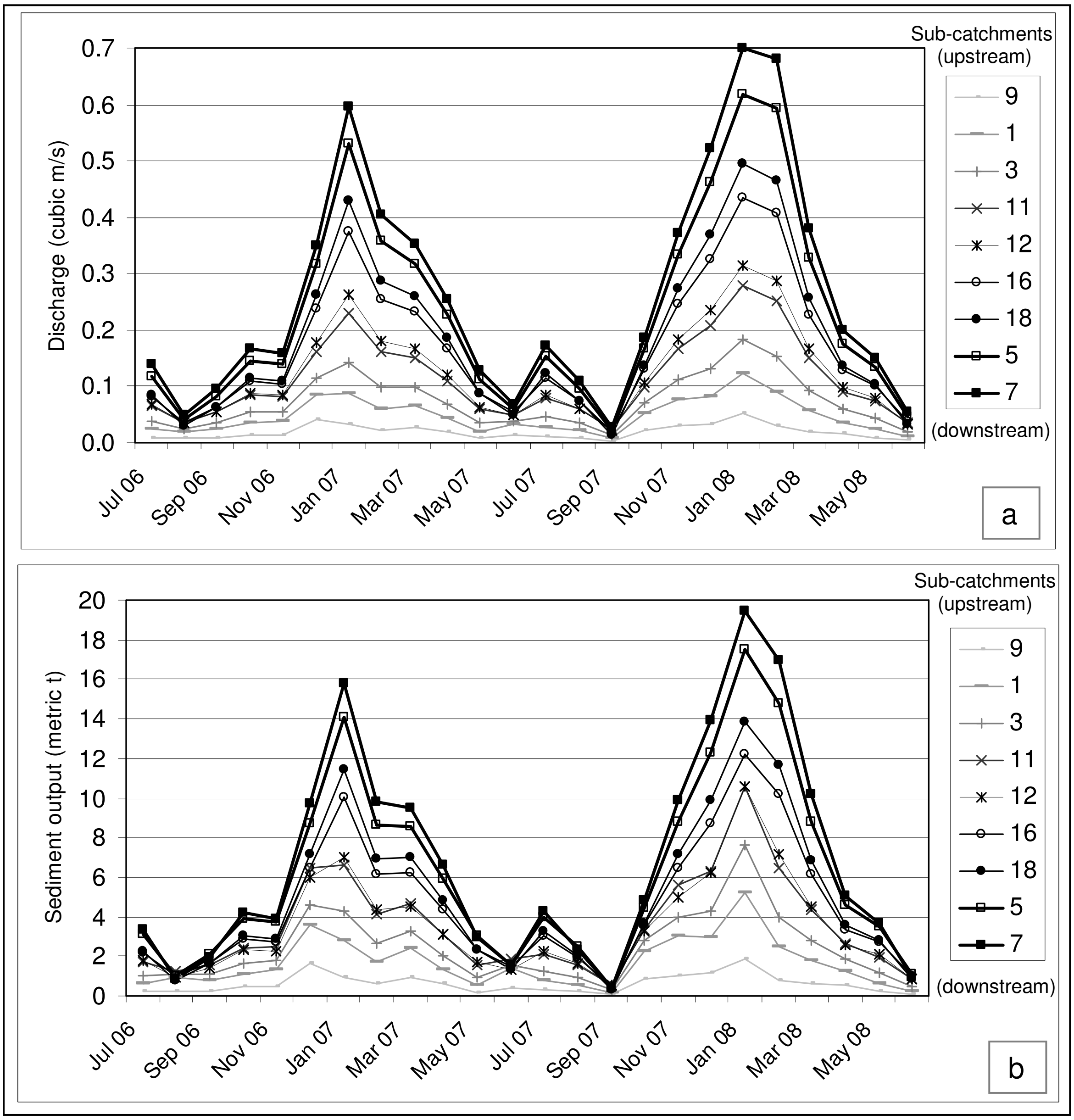

Figure 6: **(a) Monthly average streamflow (m^3/s) for 9 sub-catchments connected with the main channel; (b) Total sediment (metric t per month) transported out of the 9 sub-catchments. Sub-catchment numbers are assigned arbitrarily.**

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.

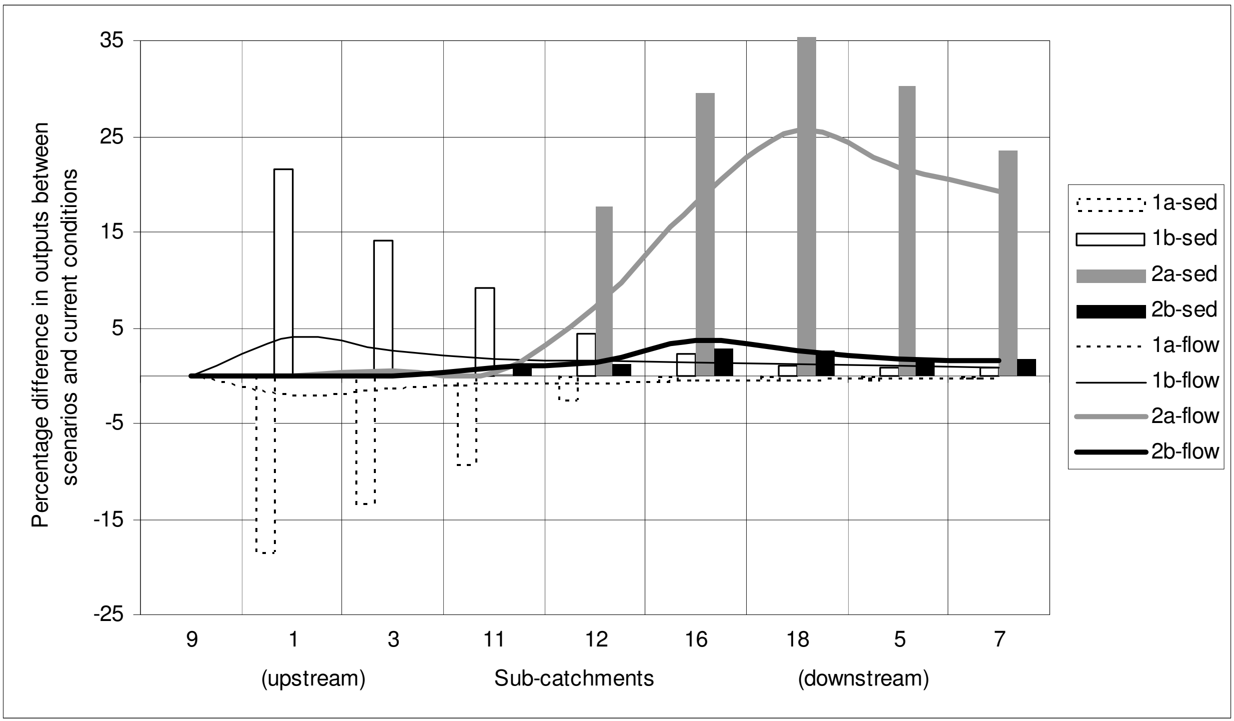

Figure 7: **Scenario impacts on discharge and sediment output for 9 sub-catchments long the main channel, expressed as the percentage difference between current conditions and four scenarios (1a - replacing cabbage plot with sugarcane; 1b – expanding cabbage plot by approximately 300%; 2a - current conditions 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.

Acknowledgements

The authors are thankful for research funding and managerial support from the Water Research Commission, especially Mr. M. Du Plessis and the Agricultural Research Council - Institute for Soil, Climate and Water. Special thanks to Prof. A. Görgens and Ms. S. Lyons at Aurecon (formerly Nimham Shand Consulting Civil Engineers) for coordination of the larger project. The study benefited from the comments from project team members including Prof. J. Annandale, Mr. M. Van der Laan, Dr. N. Jovanovic, and Dr. B. Grové and Ms. N. Matthews. Thanks also to advice from the appointed Reference Group, especially from Prof. K.L. Bristow.

References

Agrometeorology Staff. 1984-2008. *ARC-ISCW Agrometeorology Weather Station Network Data for South Africa*, *Unpublished*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Arnold JG, Srinivasan R, Muttiah RS, Williams JR. 1998. Large area hydrologic modeling and assessment part I: model development. *Journal of the American Water Resources Association* **34**(1): 73–89.

Arnold JG, Williams JR Maidment DR. 1995. Continuous-time water and sediment-routing model for large basins. *Journal of Hydraulic Engineering* **121**(2): 171-183.

Bouraoui F, Benabdallah S, Jrad A, Bidoglio G. 2005. Application of the SWAT model on the Medjerda River Basin (Tunisia). *Physics and Chemistry of the Earth* **30**: 497–507.

Bracken LJ, Croke J. 2007. The concept of hydrological connectivity and its contribution to understanding runoff-dominated geomorphic systems. *Hydrological Processes* **21**: 1749–1763.

Chen E, Mackay DS. 2004. Effects of distribution-based parameter aggregation on a spatially distributed agricultural nonpoint source pollution model. *Journal of Hydrology* **295**: 211–224.

Collins AL, Walling DE. 2004. Documenting catchment suspended sediment sources: problems, approaches and prospects. *Progress in Physical Geography* **28**(2): 159–196.

Council for Geoscience. 2007. *Geological data 1:250 000*. Council for Geoscience: Pretoria, South Africa.

De Vente J, Poesen J. 2005. Predicting soil erosion and sediment yield at the basin scale: Scale issues and semi-quantitative models. *Earth-Science Reviews* **71**: 95–125.

De Vente J, Poesen J, Arabkhedri M, Verstraeten G. 2007. The sediment delivery problem revisited. *Progress in Physical Geography* **31**(2): 155–178.

Di Luzio M, Srinivasan R, Arnold J. 2004. A GIS-coupled hydrological model system for the watershed assessment of agricultural non-point and point sources of pollution. *Transactions in GIS* **8**(1): 113-136.

Evans IS. 1979. *An integrated system of terrain analysis and slope mapping. Final report on grant DA-ERO-591-73-G0040*. University of Durham: England. http://www.soi.city.ac.uk/~jwo/phd.

Flügel W, Märker M, Moretti S, Rodolfi G, Sidrochuk A. 2003. Integrating geographical information systems, remote sensing, ground truthing and modeling approaches for regional erosion classification of semi-arid catchments in South Africa. *Hydrological Processes* **17**: 929-942.

Gassman PW, Reyes MR, Green CH, Arnold JG. 2007. The Soil and Water Assessment Tool: historical development, applications, and future research directions. *Transactions of the ASABE* **50**(4): 1211-1250.

GISCOE. 2001. *GISCOE Digital Terrain Models*, *Unpublished*. GIMS: Midrand, South Africa.

Green WH, Ampt GA. 1911. Studies on soil physics, 1. The flow of air and water through soils. *Journal of Agricultural Sciences* **4**:11-24.

Hatterman FF, Krysanova V, Habeck A, Bronstert A. 2006. Integrating wetlands and riparian zones in river basin modeling. *Ecological Modelling* **199**(4): 379-392.

Hooke J. 2003. Coarse sediment connectivity in river channel systems: a conceptual framework and methodology. *Geomorphology* **56**:79-94.

Kakembo V, Ndlela S, Cammeraat E. 2012. Trends in vegetation patchiness loss and implications for landscape function: the case of *Pteronia Incana* invasion in the Eastern Cape Province, South Africa. *Land Degradation and Development*, In press. DOI: 10.1002/ldr.2175.

Kirkby MJ, Imeson AC, Bergkamp G, Cammeraat LH. 1996. Scaling up processes and models from the field plot to the watershed and regional areas. *Journal of Soil and Water Conservation* **51**(5):391-396.

Land Type Survey Staff. 1972–2008. *Land Types of South Africa: Digital Map (1:250 000 scale) and Soil Inventory Databases*. ARC-Institute for Soil, Climate and Water: Pretoria, South Africa.

Lane LJ, Hernandez M, Nichols M. 1997. Processes controlling sediment yield from watersheds as functions of spatial scale. *Environmental Modelling and Software* **12**(4): 355-369.

Lenhart T, Van Rompaey A, Steegen A, Fohrer N, Frede H, Govers G. 2005. Considering spatial distribution and deposition of sediment in lumped and semi-distributed models. *Hydrological Processes* **19**: 785–794.

Le Roux PAL, Fraenkel CH, Bothma CB, Gutter JH, Du Preez CC. 2006. *Soil survey report: Mkabela catchment*. Department of Soil, Crop and Climate Sciences, University of the Free State: Bloemfontein, South Africa.

Le Roux JJ, Germishuyse T, Lorentz S. 2009. *Case studies and scenarios of NPS control measures and mitigation of downstream pollution: Using the Soil and Water Assessment Tool, KwaZulu-Natal. ISCW Report No. GW/A/2009/04*. ARC-ISCW: Pretoria, South Africa.

Lesschen JP, Schoorl JM, Cammeraat LH. 2009. Modelling runoff and erosion for a semi-arid catchment using a multi-scale approach based on hydrological connectivity. *Geomorphology* **109**: 174–183.

Lorentz SA, Miller J, Lechler P, Mackin G, Lord M, Kollongei JK, Pretorius J, Ngeleka K, Zondi N, Le Roux JJ. 2011. *Definition of process zones and connectivity in catchment scale NPS processes, WRC report 1808/1/11*. Water Research Commission: Pretoria, South Africa.

Martinec J, Rango A. 1989. Merits of statistical criteria for the performance of hydrological models. *Water Resources Bulletin* **25**: 421–432.

Medeiros PHA, Güntner A, Francke T, Mamede GL, De Araújo JC. 2010. Modelling spatio-temporal patterns of sediment yield and connectivity in a semi-arid catchment with the WASA-SED model. *Hydrological Sciences Journal* **55**(4): 636–648.

Mishra A, Froebrich J, Gassman PW. 2007. Evaluation of the SWAT model for assessing sediment control structures in a small watershed in India. *Transactions of the ASABE* **50**(2): 469-478.

Monteith JL. 1977. Climate and the efficiency of crop production in Britain. *Philosophical Transactions of the Royal Society of London, Series B* **281**:277-329.

Mucina L, Rutherford MC. 2006. *The vegetation of South Africa, Lesotho and Swaziland*. *Strelitzia* **19**. South African National Biodiversity Institute, Pretoria, South Africa.

Nash JE, Sutcliffe JV. 1970. River flow forecasting through conceptual models. *Journal of Hydrology* **10**: 282–290.

Neitsch SL, Arnold JG, Kiniry JR, Williams JR. 2005. *Soil and Water Assessment Tool Theoretical Documentation, Version 2005*. USDA Agriculture Research Service, Blackland Research Center, Texas Agricultural Experiment Station: Texas, USA.

Overton DE. 1966. Muskingum flood routing of upland streamflow. *Journal of Hydrology* **4**: 185-200.

Parsons AJ. 2012. How useful are catchment sediment budgets? *Progress in Physical Geography*, In press. DOI: 10.1177/0309133311424591.

Priestley CHB, Taylor RJ. 1972. On the assessment of surface heat flux and evaporation using large-scale parameters. *Monthly Weather Review* **100**:81-92.

Prosser IP, Rutherfurd ID, Olley JM, Young WJ, Wallbrink PJ, Moran CJ. 2001. Large-scale patterns of erosion and sediment transport in river networks, with examples from Australia. *Marine Freshwater Research* **52**: 81–99.

Refsgaard JC, Hansen JR. 2010. A good-looking catchment can turn into a modeller's nightmare. *Hydrological Sciences Journal* **55**(6): 849–856.

Schmidt J, Evans IS, Brinkmann J. 2003. Comparison of polynomial models for land surface curvature calculation. *International Journal of Geographical Information Science* **17**(8): 797-814.

Sharpley AN, Williams JR. 1990. *EPIC-Erosion Productivity Impact Calculator, model documentation. Technical Bulletin 1768.* U.S. Department of Agriculture, Agricultural Research Service: USA.

Srinivasan R, Ramanarayanan TS, Arnold JG, Bednarz ST. 1998. Large area hydrologic modeling and assessment part II: model application. *Journal of the American Water Resources Association* **34**: 91–101.

Srinivasan R, Zhang X, Arnold J. 2010. SWAT ungauged: hydrological budget and crop yield predictions in the upper Mississippi river basin. *Transactions of the ASABE* **53**(5): 1533-1546.

Tibebe D, Bewket W. 2011. Surface runoff and soil erosion estimation using the SWAT model in the Keleta watershed, Ethiopia. *Land Degradation and Development* **22**(6): 551-564.

Tol JJ, Le Roux PAL, Hensley M. 2010. Soil indicators of hillslope hydrology in the Bedford catchments, South Africa. *South African Journal of Plant and Soil* **27**(3): 242-251.

USDA SCS. 1972. *National Engineering Handbook, Section 4 Hydrology*. USDA Agricultural Conservation Service: USA.

Van Zyl AJ. 2007. A knowledge gap analysis on multi-scale predictive ability for agriculturally derived sediments under South African conditions. *Water Science and Technology* **55**(3): 107-114.

Walling DE, 1999. Linking land use, erosion and sediment yields in river basins. *Hydrobiologia* **410**: 223– 240.

Wang X, Yang W, Melesse AM. 2009. Using Hydrologic Equivalent Wetland Concept within SWAT to estimate streamflow in watersheds with numerous wetlands. *Transactions of the ASABE* **51**(1): 55-72.

Williams JR. 1969. Flood routing with variable travel time or variable storage coefficients. *Transactions of the ASABE* **12**(1):100-103.

Williams JR. 1975. Sediment-yield prediction with universal equation using runoff energy factor. In *Present and Prospective Technology for predicting sediment yield and sources. ARS.S-40*, US Gov. Print. Office: Washington, DC; 244–252.

Wischmeier WH, Smith DD. 1978. *Predicting Rainfall Erosion Losses, a Guide to Conservation Planning. USDA Agriculture Handbook No. 537*. USDA: Washington DC, USA.