3. CASE STUDIES

Preface

Section 3 comprises three chapters as follows:


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

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.

<table>
<thead>
<tr>
<th>Abbreviation</th>
<th>Name</th>
<th>Developed by</th>
<th>Aim</th>
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<tbody>
<tr>
<td>GLASOD</td>
<td>Global assessment of human-induced soil degradation</td>
<td>International Soil Reference and Information Centre (ISRIC) (Oldeman et al., 1991)</td>
<td>Actual soil erosion based on distributed point data obtained from various experts. Soil-erosion areas were delineated according to their judgment.</td>
</tr>
<tr>
<td>SDPM</td>
<td>Sediment Delivery Potential Map</td>
<td>Water Research Commission (WRC) (Rooseboom et al., 1992)</td>
<td>To provide spatial data on sediment yield by gathering sediment data and relevant geographical information which influences sediment yield values of catchments.</td>
</tr>
<tr>
<td>BSI</td>
<td>Bare Soil Index</td>
<td>Agricultural Research Council – Institute for Soil, Climate and Water (ARC-ISCW) (Pretorius and Bezuidenhout, 1994)</td>
<td>To detect bare soil and the status of extensive eroded areas on a national scale with Landsat Thematic Mapper (TM) data.</td>
</tr>
<tr>
<td>ESM</td>
<td>Erosion Susceptibility Sap</td>
<td>Agricultural Research Council – Institute for Soil, Climate and Water (ARC-ISCW) (Pretorius, 1995)</td>
<td>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.</td>
</tr>
<tr>
<td>PWEM</td>
<td>Predicted Water Erosion Map</td>
<td>Agricultural Research Council – Institute for Soil, Climate and Water (ARC-ISCW) (Pretorius, 1998)</td>
<td>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.</td>
</tr>
<tr>
<td>NRA</td>
<td>Natural Resources Auditing</td>
<td>Agricultural Research Council – Institute for Soil, Climate and Water (ARC-ISCW) (Wessels et al., 2001a) (Wessels et al., 2001b)</td>
<td>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).</td>
</tr>
<tr>
<td>ISRDS nodes</td>
<td>Integrated Sustainable Rural Development Strategy nodes</td>
<td>Agricultural Research Council – Institute for Soil, Climate and Water (ARC-ISCW) (Ströhmenger et al., 2004)</td>
<td>As above</td>
</tr>
<tr>
<td>SANBI</td>
<td>South African National Biodiversity Institute land degradation review</td>
<td>SANBI (Garland et al., 2000)</td>
<td>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.</td>
</tr>
</tbody>
</table>

**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.

**Rainfall erosivity (R)**

The $R$-factor is the mean annual sum of individual storm $E_{30}$ 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 $E_{30}$ 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

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\(^1\) 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.](image-url)
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 inMpumalanga, 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.

<table>
<thead>
<tr>
<th>Erosion</th>
<th>No Erosion</th>
<th>Row Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>n (&gt;10 t/ha·yr)¹</td>
<td>767</td>
<td>1 947</td>
</tr>
<tr>
<td>n (&lt;10 t/ha·yr)²</td>
<td>408</td>
<td>7 168</td>
</tr>
<tr>
<td>Column total</td>
<td>1 175</td>
<td>9 115</td>
</tr>
<tr>
<td>Omission³</td>
<td>0.65</td>
<td>0.78</td>
</tr>
<tr>
<td>Commision⁴</td>
<td>0.28</td>
<td>0.94</td>
</tr>
<tr>
<td>Total accuracy</td>
<td>0.77</td>
<td></td>
</tr>
</tbody>
</table>

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 remain 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 Système 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.

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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

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 km²) 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 30° 46' 58" and 31° 28' 55" south and 27° 55' 56" and 29° 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 km² 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 km² (70%) of the catchment area. The main land use is subsistence grazing (540 km² 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).
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 an 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. 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).

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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.

<table>
<thead>
<tr>
<th>Contributing factor</th>
<th>Description and method of derivation</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Slope (S)</strong></td>
<td>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</td>
</tr>
<tr>
<td></td>
<td>0-5.00 (1105) 5.00-10.0 (1105) 10.0-19.0 (989) 19.0-34.0 (873) 34.0-100 (852)</td>
</tr>
<tr>
<td><strong>Upslope contributing area (AS)</strong></td>
<td>Upslope area per unit width of contour (in m²) extracted from above-mentioned 20 m resolution DEMs using the D-Inf multiple flow algorithm in TauDEM</td>
</tr>
<tr>
<td></td>
<td>0-100 (1598) 100-200 (1297) 200-400 (1037) 400-800 (502) &gt;800 (462)</td>
</tr>
<tr>
<td><strong>Topographic wetness index (TWI)</strong></td>
<td>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)</td>
</tr>
<tr>
<td></td>
<td>0-0.36 (866) 0.36-0.39 (939) 0.39-0.42 (984) 0.42-0.46 (1039) 0.46-1.00 (1066)</td>
</tr>
<tr>
<td><strong>Sediment transport capacity index (LS)</strong></td>
<td>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).</td>
</tr>
<tr>
<td></td>
<td>0-1.02 (1110) 1.02-2.30 (1080) 2.30-3.98 (976) 3.98-6.85 (885) 6.85-12.6 (874)</td>
</tr>
<tr>
<td><strong>Terrain unit (TU)</strong></td>
<td>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)</td>
</tr>
<tr>
<td></td>
<td>Crest (351) Convex midslope (2284) Concave midslope (2062) Footslope (178) Valley floor (87)</td>
</tr>
<tr>
<td><strong>Geology type (GT)</strong></td>
<td>Stratigraphic/lithologic polygon descriptions at a 1:250 000 scale (Council for Geoscience, 2007)</td>
</tr>
<tr>
<td></td>
<td>Drakensberg basalt, Karoo dolerite (777) Elliot mudstones, subordinate sandstone (779) Molteno sandstones (1571) Alluvium, mudrock, fine-grained sandstone (595) Tarkastad mudstones (1204)</td>
</tr>
<tr>
<td><strong>Land type (LT)</strong></td>
<td>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)</td>
</tr>
<tr>
<td></td>
<td>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)</td>
</tr>
<tr>
<td><strong>Soil erodibility factor (K)</strong></td>
<td>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)</td>
</tr>
<tr>
<td></td>
<td>0-0.20 (367) 0.20-0.25 (588) 0.25-0.30 (1530) 0.30-0.35 (1564) 0.35-0.70 (871)</td>
</tr>
<tr>
<td><strong>Soil depth (SD)</strong></td>
<td>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)</td>
</tr>
<tr>
<td></td>
<td>Shallow (813) Medium (2140) Deep (1330) n.a. n.a.</td>
</tr>
<tr>
<td><strong>Land use (LU)</strong></td>
<td>National Land Cover database of South Africa derived from Landsat TM imagery with a grid cell resolution of 30 m (National Land Cover, 2000)</td>
</tr>
<tr>
<td></td>
<td>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)</td>
</tr>
<tr>
<td><strong>Vegetation cover (VC)</strong></td>
<td>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)</td>
</tr>
<tr>
<td></td>
<td>0-20.0 (897) 20.0-30.0 (987) 30.0-40.0 (1115) 40.0-50.0 (928) 50.0-100 (903)</td>
</tr>
</tbody>
</table>

(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.

<table>
<thead>
<tr>
<th>Type</th>
<th>Count</th>
<th>Area (ha)</th>
<th>Gullied area (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Continuous</td>
<td>236</td>
<td>2 905</td>
<td>55</td>
</tr>
<tr>
<td>Discontinuous</td>
<td>4 017</td>
<td>2 368</td>
<td>45</td>
</tr>
</tbody>
</table>

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.](image-url)
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 m²). 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-IS CW) 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-IS CW, 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.

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Africa.

Gully and sheet erosion on subtropical mountain slopes: Their respective roles and the scale effect. 


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

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 subcatchments 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 29° 21' 12" and 29° 27' 16" south and 30° 36' 20" and 30° 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 41 54 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
The 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 21°C and 17 to 28°C, 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.](image)

**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 km²) 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).

<table>
<thead>
<tr>
<th>Sub-catchment</th>
<th>Dam area (ha)</th>
<th>Dam volume (m³)</th>
<th>Wetland area (ha)</th>
<th>Wetland volume (m³)</th>
</tr>
</thead>
<tbody>
<tr>
<td>5</td>
<td>-</td>
<td>-</td>
<td>5.44</td>
<td>108 800</td>
</tr>
<tr>
<td>10</td>
<td>0.7</td>
<td>11 800</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>11</td>
<td>5.9</td>
<td>229 600</td>
<td>4.73</td>
<td>141 900</td>
</tr>
<tr>
<td>12</td>
<td>4.5</td>
<td>87 000</td>
<td>9.17</td>
<td>183 400</td>
</tr>
<tr>
<td>13</td>
<td>1.7</td>
<td>31 800</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>14</td>
<td>8.4</td>
<td>330 400</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>15</td>
<td>1.5</td>
<td>26 600</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>16</td>
<td>10.3</td>
<td>405 600</td>
<td>22.44</td>
<td>673 200</td>
</tr>
<tr>
<td>17</td>
<td>1.2</td>
<td>20 400</td>
<td>4.88</td>
<td>97 600</td>
</tr>
<tr>
<td>19</td>
<td>2.5</td>
<td>47 800</td>
<td>-</td>
<td>-</td>
</tr>
</tbody>
</table>

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.
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:

$$Dv = \left[ \frac{V - V'}{V} \right] \times 100$$

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.
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.

Figure 4: Observed and simulated discharge and sediment loads of 5 rainfall events that occurred from October 2007 to January 2008.
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 m³/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 m³/s at sub-catchment 1 in September 2007 to 0.701 m³/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.
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.
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.

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