AN ASSESSMENT OF THE HYDROLOGIC RESPONSE OF THE KEISKAMMA CATCHMENT TO LAND USE/COVER CHANGES

BY

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DECLARATION

I, **MLAMLA SIVE,** solemnly declare that this thesis was compiled and written by me. It has never been presented anywhere for any academic award or published in any peer reviewed journals. Therefore, all materials used from other sources are duly appreciated and properly acknowledged.

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ABSTRACT

The Keiskamma catchment has undergone significant land use/cover changes (LUCC) underpinned by land use policy reforms and climate change. However, the hydrological responses of the catchment to LUCC are not fully understood.

This study sought to assess the hydrological response of the Keiskamma catchment to LUCC at catchment and hillslope scale using remote sensing, GIS, hydrological modelling and field experiments. Catchment scale assessments first involved LUCC mapping in IDRISI TerrSet software, using supervised image classification for two sets of multispectral imagery; namely Landsat Thematic Mapper (TM) of 1994 and Landsat 08 Operational Land Imager (OLI) of 2016. The LUCC maps provided an indication of LUCC over time and were prerequisite land use inputs for modelling the hydrologic response of the catchment. The Soil and Water Assessment Tool (SWAT) hydrologic model was used to model the hydrologic response of the catchment to LUCC. The Sequential Uncertainty Fitting (SUFI-2) in SWAT-CUP was used to assess model performance and uncertainty analysis. The influence of rainfall on the hydrologic response of the catchment was also assessed using linear regression. One of the prominent forms of LUCC in the Keiskamma catchment, particularly central Keiskamma is *P. incana* shrub encroachment. Field experiments were set up to investigate the hydrologic impacts of *P. incana* shrub invasion at hillslope scale, as well as to validate the results obtained by the SWAT hydrologic model. Field experiments included an assessment of the Landscape Organisation Index (LOI) of the invasion, as well as assessing of surface conditions, surface runoff (L), volumetric soil water content (cm^3/cm^3) and sediment loss (grams) under *P. incana*, grass and bare-eroded areas.

High image classification accuracy assessment values of 87.2 % and 87.4 % for 1994 and 2016 respectively were obtained, with a Kappa coefficient of 0.84 for both sets of imagery. Results of the study revealed a significant increase in woody vegetation encroachment, specifically shrub invasion, forest expansion in the upper parts of the catchment, as well as an increase in exotic and invasive vegetation species within the riparian zone. The SWAT model showed a good (NSE=0.69, R^2=0.69 and RSR =0.56) and unsatisfactory (NSE=0.4, R^2=0.4 and RSR 0.79) model performance for calibration and validation respectively. However, for both the calibration (p-factor =0.77; r-factor 1.03) and validation (p-factor =0.92; r-factor 1.38) periods

there was acceptable uncertainty as indicated by the p- and r-factor statistics. The mean annual streamflow (-71.4 %), surface runoff (-98.8 %), soil water content (-4.5 %), evapotranspiration (-5.3 %), groundwater (-79.5 %) and sediment loss (-99.9 %) decreased from 1994 to 2016. The impoundments in the catchment *viz* Cata, Mnyameni, Binfield, Sandile, Debe and Dimbaza dams, also contributed significantly to the streamflow reduction. A strong correlation (r= 0.61) between the declining streamflow (m^3/s) and rainfall (mm) was observed. At hillslope scale, *P. incana* invasion was characterised by a low LOI, owing to large inter-shrub bare patches and poor soil surface conditions characterised by soil surface crusting, conducive to high runoff generation and connectivity. High surface runoff and soil losses were evident under *P. incana* and bare-eroded areas. Volumetric soil water content was high under grass and *P. incana* tussocks, intermediate in *P. incana* inter-patches and low in bare-eroded areas. The findings and analysis of this study conclude that the hydrologic response of the Keiskamma catchment was influenced significantly by LUCC in the form of extensive invader shrub encroachment, expansion of forestry using exotic tree species, impoundments, as well as the infestation of riparian zones by invasive vegetation. Management of woody shrub encroachment and alien invasive plants as well as indigenous forest species utilisation should be considered as amongst the key efforts towards restoring the ecohydrological integrity of the Keiskamma catchment.

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1 CHAPTER 1: INTRODUCTION

1.1 General introduction

Globally, catchment areas are undergoing perpetual environmental and anthropogenic driven changes. The ongoing changes or modifications of catchment areas hold serious implications for catchment hydrological responses (Gyamfi *et al.*, 2016). Catchment hydrological response refers to the behaviour of hydrological processes in response to rainfall events, physical and geomorphological changes within a catchment area (Morán-Tejeda *et al.*, 2015). Hydrological response is a function of several factors such as climate, rainfall characteristics, watershed morphology, basin physiography and catchment hydrographic system (Bingner, 2000). These factors have fundamental control on the hydrological response of a catchment area (Cui *et al.*, 2012; Wang *et al.*, 2012b; Martins *et al.*, 2016).

According to Barasa *et al.* (2017), hydrological response is closely related to spatial distribution of the hydrological cycle processes, and tends to vary over temporal and spatial scales. de Araujo and Piedra (2009) observe that catchments can be of fairly similar size and characteristics but differ in hydrological response owing to differences in climatic conditions. Gallart *et al.* (2002) also point out that humid catchments have two times higher rainfall readings compared to their arid counterparts. In a catchment in a humid setting, discharge (and surface water availability that is five and 14 times more than that of semiarid catchments was measured, respectively (Domingo *et al.*, 2001). The findings highlight that the climatic environment within which a catchment exists, inter alia, primarily determines the hydrological response of a particular catchment.

Over the years, it has been observed that catchment hydrological response is sensitive to significant land use/cover changes over spatial time scales (Ellis and Pontius, 2006; Arsanjani, 2012). Land cover refers to the biological and physical aboveground cover components of the land surface such as vegetation, bare soil and water (Ellis and Pontius, 2006; Arsanjani, 2012). Land use is defined as anthropogenic activities that the land is utilised for such as agriculture, commercial forestry and man-made structures (Butt *et al.*, 2015). Land use/cover change is further defined as transformation of the earth’s surface by natural processes and anthropogenic modifications.

Land use/cover changes have been prominently driven by urbanisation, agricultural growth, vegetation changes and land degradation (Odindi *et al.*, 2012a). Several studies have shown that land use/cover changes induced by increased urbanisation, agricultural development and deforestation are central to changes of the water balance and catchment hydrologic response (Legesse *et al.*,2003; Cui *et al.*, 2012; Dos Santos *et al*., 2014; Morán-Tejeda *et al*., 2015; Gyamfi *et al*., 2016; Woldesenbet *et al.*, 2017).

Expansion of agriculture and urban development were found to have profound impacts on the hydrologic response of the Oliphant's catchment, in Limpopo Province, South Africa (Gyamfi *et al*., 2016). Over a period of 13 years Gyamfi *et al.* (2016) reported that the Oliphant's catchment has undergone a 31.6 % decline of rangelands with a notable increase in agriculture (20.1 %), built-up areas (10.5 %) and 0.7 % of forests. These changes resulted in a significant increase of surface runoff generation by 46.7 %. Urban development was the biggest contributor to surface runoff conditions (Gyamfi *et al*., 2016). Cui *et al.* (2012) indicated that deforestation induces a significant increase of catchment water yield owing to reduced interception and evapotranspiration. Conversely, Legesse *et al.* (2003) showed that the conversion of cropland to woodland results in an 8% decrease of streamflow.

Hydrological impacts of urbanisation are profound in urban catchments whereas subdued in rural catchment areas. Vegetation changes critically modulate the hydrological response of rural catchments. Vegetation destruction directly impacts the hydrological response (Sun *et al.*, 2017a). Uncontrolled invasion of Invasive Alien plant species (IAPs) in rural catchments have negative impacts on the catchment's hydrological response compared to native species. Dzikiti *et al.* (2013) highlighted that IAPs have great impacts on streamflow reduction, lowering of groundwater reserves, reduction of grazing land, loss of biological diversity, exacerbation of dramatic wild fires and severe soil erosion. Odindi & Kakembo (2011) showed that areas invaded by *Pteronia incana (P. incana)* were inherent of soil moisture deficit compared to native grassland which maintained significantly high soil moisture content. The patchy distribution of the invader and the bare inter-patches have implications for hydrological responses (Kakembo *et al.*, 2007; Kakembo, 2009; Odindi & Kakembo, 2011; Manjoro *et al.*, 2012; Odindi *et al.*, 2012).

To influence catchment management policy and decision-making on managing the hydrological impacts of land use/cover change, scientific involvement is important. Remote Sensing in conjunction with Geographic Information Systems are essential tools of mapping and quantifying hydrological impacts of land use/cover change (Reis, 2008; Mhangara, 2011; Dos Santos *et al.,* 2014; Butt *et al.,* 2015). Field experiments also play a critical role in highlighting the hydrological response of catchments to changes in land use and land cover. However, field experiments are limited to hillslope spatial scale. To obtain catchment scale hydrological responses information, hydrological models become important. Hydrological models use mathematical equations and natural laws to simulate and predict natural processes. Despite the usefulness of hydrological models, they are not free of bias. Nevertheless, hydrological models had proved to be successful to mimic and predict reality processes to inform decision-making in catchment resources management (Xu, 2002; Devia *et al.*, 2015).

Hydrologic models are useful tools for assessing the hydrologic response of a catchment to land use/cover changes. Koneti *et al.* (2018) used the Hydrologic Engineering Centre-Hydrologic Modelling System (HEC-HMS) hydrologic model, Remote Sensing and GIS techniques to investigate the spatial and quantitative changes in the hydrologic response arising from land use/cover changes between 1985 and 2014 in the Godavari River Basin, India. Koneti *et al.* (2018) highlighted that integrating GIS, Remote Sensing and the HEC-HMS hydrological model can solve certain hydrological related problems in river basins. A similar investigation by Zhang *et al.* (2014) examined the hydrological impacts of land use/cover change using the Variable Infiltration Capacity (VIC) model and found that such an approach can be useful for water management experts. In Nigeria, Adeogun *et al.* (2014) used the Soil and Water Assessment Tool (SWAT) hydrologic model to predict water yield and water balance. SWAT proved to be a promising tool for predicting the impacts of land/use cover change on water resources management (Adeogun *et al.*, 2014; Gyamfi *et al.*, 2016; Sun *et al.*, 2017b) Therefore, studying catchment hydrologic response also supports water resource planning and conservation. Accurate hydrologic response determination is fundamental for catchment water yield estimations and predictions. Water availability forecasts are also an integral part of integrated water resources management.

Understanding of catchment hydrologic responses is also important for long-term insights into the impacts of future climate and land use change scenarios in water resources (Devia *et al.*, 2015; Gyamfi *et al.*, 2016).

1.2 Problem statement

The Keiskamma catchment is highly vulnerable to land use/cover changes. The land use/cover changes are fuelled by climatic, ecological and anthropogenic factors. Keiskamma catchment was found predisposed to vegetation stress (Haindongo, 2009). Mhangara *et al.* (2011) identified vegetation degradation and found the catchment prone to land degradation. The observed land degradation is explained to have led to subsequent severe soil erosion and high sediment yields (Mhangara, 2011). The high soil erosion rates were reported to be perpetuated by degraded vegetation and bare surface conditions coupled by underlying problem soils. Mhangara *et al.* (2011) identified high sodium content and low concentrations of soluble salts which promote soil dispersion and subsequently lead to piping and gully erosion. In addition, Mhangara *et al.* (2011) demonstrated that intact vegetation had declined significantly from 1972-2006.

The findings by Haindongo (2009) and Mhangara *et al.* (2011) have implications for the hydrological response of the catchment. However, previous studies conducted in the Keiskamma catchment have not assessed the hydrologic impacts of land use/cover changes over time. Therefore, the present study seeks to determine the hydrologic impacts of land use and land cover changes in the Keiskamma catchment.

1.3 Aim of the study

The aim of the study was to determine the hydrological impacts of land use/cover change in the Keiskamma catchment and to generate baseline hydrologic response information that is, inter alia, useful for land rehabilitation and water resources management, as well as long-term insights into the impacts of future climate and land use change scenarios in water resources within the catchment.

1.4 Specific objectives

- **To assess land use/cover changes of the Keiskamma catchment for 1994 and 2016.** – Multispectral imagery from the LANDSAT programme were used to assess land use/cover changes in the catchment. An assessment of the land use/cover changes is important to track what has changed over time and for deriving inferences for explaining links between the changes and hydrological responses. The land use maps produced were used as inputs for modelling the hydrologic response of the Keiskamma catchment.

- **To model the hydrologic response of the Keiskamma catchment to land use/cover changes from 1994 to 2016**. To achieve this, the Soil and Water Assessment Tool (SWAT) hydrologic model was used to model hydrologic responses of the catchment to land use/cover changes. The model was set up using all the required datasets *viz* weather, streamflow, soil and land use/cover data. The two land use/cover change maps, generated from objective one, were used as inputs to the model during calibration (1994-2000) and validation (2010-2016). The modelled hydrology of the catchment is a critical indicator of hydrological conditions of the catchment. Land use/cover change was correlated with the simulated hydrological components to assess the land use/cover change-hydrology relationship. Changes in land use/cover were used to explain the hydrologic response over the years.

- **To determine the hydrological impacts of invader shrubs in the catchment, particularly *Pteronia incana*.** Field experiments were setup in *P. incana* invaded hillslopes to assess the hydrological responses in the invaded areas. Findings of the field experiments can permit an extrapolation to other similarly invaded catchments as well as provide validation for simulations of the SWAT hydrologic model.

1.5 Research questions

- **Research Question 1:** How has land use/cover of the Keiskamma catchment changed over the 22-year study period?
- **Research Question 2:** What is the impact of the individual land use/cover changes on the hydrological response of the catchment?
- **Research Question 3:** What are the hydrological impacts of *Pteronia incana* invasion in the Keiskamma catchment?
- **Research Question 4:** What is the influence of rainfall variability in the hydrology of the Keiskamma catchment over the years?
- **Research Question 5:** What are the implications of the land use/cover changes and hydrological trends for rehabilitation and water resource management of the catchment?

1.6 Rationale

The present study seeks to investigate the hydrologic impacts of land use/cover changes in the Keiskamma catchment to provide baseline hydrological response information for catchment management purposes. Assessing the hydrological response of the catchment is imperative as a contribution towards understanding the relationship between land use/cover changes and the hydrological response.

The study assesses the hydrological response of the catchment to land use/cover changes at catchment and hillslope scale. The importance of investigating the changes across the two spatial scales is critical. An investigation at hillslope scale is important to highlight the hydrological response for the entire catchment. This will inform catchment management decisions that are made based on findings pertinent to the general catchment.

The investigation at hillslope scale focuses on specific aspects of land use/cover changes. At hillslope scale an investigation of the hydrological response of the catchment to the invasion by *Pteronia incana (P. incana*), commonly known as Blue Bush or "Bhosisi" was conducted. Land use/cover change dynamics prominent in invaded catchments entail the replacement of indigenous grassland by *P. incana* and land predisposition to soil erosion. Hydrological responses resulting from *P. incana* invasion on catchments are fundamental indicators of the state of rangeland condition. The hydrological response is also important for providing insights into rehabilitation of degraded land.

Furthermore, water provision in South Africa is critically based on conservation of catchment areas, wetlands and riparian zones (Marais *et al.*, 2004). According to Finca (2011), South African Water Management Areas (WMA) and catchments thereof are predominantly grassland. Marais *et al.* (2004) point out that grassland and resident vegetation in watersheds play a critical role improving water resources. It is explained that vegetation can function as a sponge that captures and retains rainfall during summer and then during the dry season the infiltrated rainfall gets gradually fed into streamflow via baseflow (Wyk, 1987; Finca, 2011). The impairment of catchment grassland and native vegetation cover disrupts such important hydrological balance processes.

Assessing hydrological impacts of land use/cover transformation is important as part of the support for management practices that seek to restore catchment ecohydrological integrity.

Another critical aspect of the study is the development of a model which can be used for future studies of water yield estimations of the catchment to inform water conservation plans and strategies. Sun *et al.* (2017) observe that "vegetation cover management is one of the most effective and sustainable methods of improving water resources in water-constrained regions."

1.7 Chapter outline

Chapter 1: Introduction

Chapter 1 of this investigation entails the introduction, problem statement, aim of the study, specific objectives, chapter outline, and gives the significance of the investigation. The introduction focuses on an overview of the topic of investigation. It highlights land use/cover changes causes, hydrological responses and controls of hydrological responses.

Chapter 2: Literature Review

First, the literature review of this study focuses on discussing catchment hydrological processes. Second, the theoretical background on hydrological modelling is discussed. The section also describes the selected model, the SWAT and its accuracy assessment. Thirdly, the literature review discusses the role of climate, vegetation and land use changes on the hydrological response of the catchment. This is basically, a review of factors regulating catchment hydrological responses. A review is also made of methods of qualitatively and semi-quantitatively determining catchment hydrological responses using field methods and techniques.

Chapter 3: Study Area

Firstly, this chapter presents the criteria for site selection for this study. The chapter further describes the catchment in terms of its geographic location, channel network, climate, vegetation, soils, geology, topography as well as demography and land use.

Chapter 4: Methods and Materials

The chapter presents the methods and techniques used for this investigation. Firstly, the chapter details the process of satellite imagery classification using a pixel-based supervised method. The section also details the accuracy assessment process of the classified images as a way to validate the degree of reliability of the produced land use maps for decision-making. The maps were then further used as land use map inputs for the SWAT model. Secondly, the process of modelling the catchment hydrological response using the SWAT hydrologic model is explained. Model calibration and validation are also explained concurrently with the model accuracy assessment. An assessment of the impacts of impoundments and rainfall influence on the catchment hydrological response is also presented. Thirdly, field experiment methods used to determine hydrological impacts of *Pteronia incana* are outlined.

Chapter 5: Results

This chapter provides findings of this study. The first section presents result of accuracies of classified maps of 1994 and 2016. After this, classified land use/cover change maps of the Keiskamma catchment are presented. Quantitative results of land use/cover changes from 1994 to 2016 also feature in this section.

The second section presents findings of the SWAT model performance assessment for the calibration and validation period. After this, hydrological responses of the catchment for 1994 and 2016 land use/cover scenarios are presented. The impacts of impoundments on streamflow are also presented. Results of the influence of rainfall variability on the hydrologic response of the catchment are presented as well.

The third section of the results presents field experiment results of hydrological implications and impacts of *P. incana* invasion. The findings include hydrologic implications of landscape organisation of invaded areas as well as surface conditions, surface runoff, soil moisture content and sediment loss in invaded areas, grassland and bare-eroded hillslopes. A presentation of results of sediment loss from hillslope scale monitoring and SWAT model simulations is given as a form of validating the model.

Chapter 6: Discussion

This chapter first discusses results of land use/cover changes of the Keiskamma catchment. The impacts of land use/cover changes on the hydrological responses simulated by the SWAT hydrologic model are also discussed. A discussion of the SWAT hydrologic model's performance is also presented. The influence of rainfall variability on the hydrology of the Keiskamma catchment is given. Catchment hydrological responses owing to *P. incana* shrub invasion are discussed as well. Furthermore, the chapter provides a critique of gaps and limitations of the SWAT hydrologic model.

Chapter 7: Conclusion and Recommendations

The chapter provides an overall conclusion on land use/cover changes, the modelled hydrological responses as well hydrological impacts of *P. incana* invasion in the Keiskamma catchment. Recommendations for future investigations are made. Implications of the study findings for catchment management are also detailed.

2 CHAPTER 2: LITERATURE REVIEW

Introduction

First, a literature review of this study focuses on discussing the concept of catchment hydrological responses. Second, a review is presented of the role of climate, vegetation and land use change on catchment hydrological responses, and as factors regulating catchment hydrological responses. Third, the theoretical background on hydrological modelling is presented. In this section, an explanation of the SWAT model, which is used in this study, is given. Its calibration, validation and accuracy assessment are also highlighted. A review of methods of qualitatively and semi-quantitatively determining catchment hydrological responses using field experiments and techniques is also provided.

2.1 Catchment hydrological responses

Hydrological responses refer to the hydrological processes that occur within a catchment. Hydrological response processes include behaviour or the response of hydrologic processes such as streamflow, surface runoff, lateral flow infiltration, soil moisture fluxes, groundwater recharge and evapotranspiration. According to Zhang *et al.* (1999), catchment hydrological responses and associated changes can be evaluated using the water balance concept.

Catchment water balance is given as:

$$P = ET + R + D + \Delta S$$

Where P is precipitation, ET is evapotranspiration, R is surface runoff measured as streamflow, D for recharge to groundwater and ΔS is the change in soil water storage (Zhang *et al*., 1999, p4).

Precipitation is the source of water into a catchment. Major water pathways include surface runoff which is generated when the rate of infiltration exceeds soil water retention capacity. A portion of the precipitation that reaches the ground is stored as soil moisture, and in locales of sufficient hydraulic conductivity, infiltration recharges catchment groundwater reserves. Surface runoff feeds into streamflow. Basal flow from groundwater also contributes to streamflow. Water is lost as evaporation from water bodies and bare surfaces and as evapotranspiration (Gassman *et al.*, 2007; Owuor *et al.*, 2016; Woldesenbet *et al.*, 2017).

An ideal water balance state is where catchment land use/cover surface conditions function to conserve conditions of sufficient surface runoff to contribute to streamflow and where substantial amounts of infiltration and groundwater recharge are maintained (Cullis *et al.*, 2007). It is of utmost importance for catchment areas to maintain this state of quasi equilibrium of the water balance components. The equilibrium can be achieved through judicious management of catchment surface conditions of land use/cover types. The water balance dynamics are critically affected by catchment land use/cover change (Zhang *et al.*, 1999; Domingo *et al.*, 2001; Adeogun *et al.*, 2014).

Hydrological responses of catchments are however locale specific and are determined by several factors that include but are not limited to climate, land use, watershed characteristics and vegetation. Noteworthy is that hydrological responses also vary over temporal scales (Gyamfi *et al.*, 2016; Barasa *et al.*, 2017). Factors affecting hydrological responses of catchments are discussed in the following sections.

2.2 Hydrological responses to land use/cover changes

Land use/cover changes have a strong influence on changes in hydrological responses. In arid and semi-arid regions, land cover predominantly refers to the surface conditions inclusive of vegetation cover and topsoil conditions. Unlike urban catchments, the hydrological responses of most rural catchment areas are not affected by the impacts of urbanisation. The effects of artificial impervious surfaces are subdued rural catchments (Gallart *et al.*, 2002; Wilcox and Thurow, 2014; Gyamfi *et al.*, 2016). Against that background, this section of the literature review will focus on alterations of hydrological responses in rural catchments. This approach contextualises background information relating to conditions of the catchment under investigation, which is also in a predominantly rural setting.

Hydrological response in catchment areas is critically influenced and controlled by vegetation. Sun *et al.* (2017) used the SWAT hydrologic model to study the impacts of vegetation dynamics on the hydrological processes of the Xilin River Basin in a semi-arid steppe of Northern China. They noted that the impacts of vegetation change on catchment hydrological processes vary across different spatial and temporal scales.

Vegetation regulates catchment water infiltration, groundwater recharge, baseflow, streamflow, evapotranspiration and surface runoff. The relationship between vegetation and catchment hydrological processes is also a function of a catchment's topography, climate and soil properties (Schiariti, 2012; Barasa *et al.*, 2017; Sun *et al.*, 2017).

Vegetation undoubtedly affords soil a protective cover against high intensity rainfall and erosion. Generally, vegetation functions to bind soil particles into a firm cohesive state and enhances soil water retention capacity (Visser *et al.*, 2004; van der Maarel, 2005; Wang *et al.*, 2012b; Li *et al.*, 2016). Vegetation cover determines the amount of water that gets recharged into the ground and contributed to streamflow as overland flow and baseflow (Sun *et al.*, 2017b). This role of vegetation is also controlled by topography. Conventionally, high vegetation cover is inherent of high infiltration and subsequent groundwater recharge (Owuor *et al.*, 2016; Sun *et al.*, 2017a). This effect of vegetation in promoting soil infiltration is enhanced in gentle topography and maximised in catchment concavities and bottomlands. Conversely, in steeper and convex terrain, infiltration and groundwater recharge are not as pronounced compared to their counterparts (Hickin, 1984; Sun *et al.*, 2017a).

Catchment soil moisture is also influenced by vegetation. Well vegetated catchments are inherent of high soil moisture compared to stripped uncovered landscapes. Vegetation primarily facilitates high rates of infiltration and subsequent high water storage. Water loss in vegetated catchments is via evapotranspiration induced by incoming insolation. Nonetheless, resident vegetation reduces the effects of temperature in inducing evaporation; hence, high soil moisture contents characterise vegetated catchments (Moran *et al.*, 2009; Wang *et al.*, 2012c).

Topography also plays a critical role in regulating soil moisture content. Well vegetated gentle and concave catchment hillslopes are characteristic of high soil moisture (Kakembo *et al.*, 2007, 2009). In contrast, convex slopes generally have poor vegetation cover and low soil moisture retention (Kakembo *et al.*, 2007, 2009). This is attributed to their thin soils which do not have the capacity to anchor vegetation and provide maximal water holding capacity.

Convex topography are run-off areas and devoid of dense vegetation cover; hence reduced soil water retention capacity (Kakembo *et al.*, 2007, 2009; Lu *et al.*, 2011; Odindi and Kakembo, 2011; Sun *et al.*, 2017).

According to Finca (2011), South Africa’s Water Management Areas (WMA) are predominantly grassland. Owuor *et al.* (2016) observe that grassland and resident vegetation capture and retain rainfall during the wet season and gradually release it into streamflow via baseflow during the dry season. At catchment scale, the control of grassland and native vegetation in a catchment holds critical importance for catchment water balance, water yield as well as water quality (Eldridge *et al.*, 2015; Gyamfi *et al.*, 2016).

Sun *et a.l* (2017) point out that research on the impacts of terrestrial vegetation cover on hydrological responses is key to the development of sustainable water management strategies. Moreover, vegetation dynamics at catchment are not solely drivers of catchment hydrological behaviour. The impacts of vegetation dynamics on the hydrologic response of catchments is intertwined with other factors as explained in the sections below.

2.2.1 Climate intertwined with vegetation as controls of catchment streamflow

Climate is a critical determinant of water balance and availability in catchment areas. Climate implications for catchment hydrologic responses are manifold and are intertwined with other factors that affect hydrological processes (Legesse *et al.*, 2003; Tadele and Förch, 2007; Morán-Tejeda *et al.*, 2015; Odiyo *et al.*, 2015).The most pronounced impacts of climate change are elevated temperatures and erratic rainfall climatic conditions (Legesse *et al.*, 2003). In arid and semi-arid areas, including South Africa, prolonged drought conditions have been experienced because of climate change. The dry conditions have severely affected catchment water balance and availability as well as interlinked processes.

Reduced streamflow and baseflow contributions have been quantified (Dos Santos *et al.*, 2014; Sun *et al.*, 2017). Streamflow reduction is a function of low rainfall conditions, high evaporation and evapotranspiration rates as well as seasonality. Sun *et al.* (2017) investigated the hydrological effects of vegetation cover reduction and found that a decline in streamflow is more pronounced during the dry season.

Further, Sun *et al.* (2017) revealed that reduction of vegetation cover favours high streamflow, particularly during the growing seasons (i.e. rainfall seasons). From their study it can be deduced that catchment streamflow response is a function of catchment vegetation conditions, season of the year, and is chiefly determined by rainfall. Similarly, Dos Santos *et al.* (2014)

used SWAT to model hydrologic impacts of land use and land cover changes. Their study revealed that streamflow increases during the rainy season and substantially declines during the dry season; thus demonstrating the effects of seasonality.

Dos Santos *et al.* (2014) explained the response of streamflow as mainly due to land degradation involving the conversion of natural vegetation into livestock grazing veld. They also noted that clearing of vegetation facilitated increased surface runoff during the raining season and led to dry seasons with reduced baseflow and streamflow. The lack of catchment vegetative cover fuels high volumes of surface runoff that feed into streamflow. It, however, impedes surface infiltration and groundwater recharge. This is essentially caused by degradation of catchment vegetation cover and poor catchment surface conditions induced by malpractices such as overgrazing. Dos Santos *et al.* (2014) also highlight the importance of catchment native vegetation cover in maintaining catchment hydrological balance. Vegetation retains part of catchment water as soil moisture and to some extent as groundwater. Degradation of catchment vegetation and introduction of poor surface conditions disrupt such important water balance in catchment areas. The disruption of these critical hydrological processes leads to the deterioration of long-term water resources availability.

2.2.2 Impacts of vegetation cover dynamics on evapotranspiration (ET)

Evapotranspiration is defined as the loss of water from the soil as evaporation and from plants as transpiration through the plant. According to Zhang *et al.* (1999) and Sun *et al.* (2017), ET is one of the essential components of the catchment water balance and it is closely linked to characteristics of vegetation. Thus, evapotranspiration (ET) can be used as an indicator of catchment vegetation and surface conditions. This is supported by Cui *et al.* (2012) who state that ET is linked to climate; that is, energy and water availability and vegetation cover.

Generally, ET is higher in forests than in bare or non-forested catchment areas. Subsequently, low ET values suggest lack of vegetation cover and soil moisture whilst high ET values are indicative of vegetated and water unlimited catchments (Domingo *et al.*, 2001; Lu *et al.*, 2011; Wang *et al.*, 2012c). According to Cui *et al.* (2012), at catchment scale evapotranspiration provides essential information for understanding the water balance and water yield in catchment areas.

This can be attributed to ET as a function of vegetation, and vegetation holds profound modulating effects on hydrological responses. Cui *et al.* (2012) also showed that ET estimations of 1 mm.day^{-1} and 2-3 mm.day^{-1} were quantified for the dry and wet season, respectively. These findings suggest that owing to minimal plant activity and low rainfall events in winter, low ET values are measured. Conversely high plant activity in the wet season explains relatively higher measured ET. The study therefore demonstrated the close relation between vegetation and ET as well as the influence of seasonality.

Sun *et al.* (2017) assessed the relationship between ET and the Normalised Difference Vegetation Index (NDVI) and detected a perfect positive correlation. Where there is high vegetation cover, high ET values were quantified. Sun *et al.* (2017) also noted a significant increase in alpine vegetation and ET values. However, no significant response was found in lower altitudes (Sun *et al.*, 2017; Dzikiti *et al.*, 2014; Morán-Tejeda *et al.*, 2015). It was explained that mountainous areas had high moist conditions and vegetation cover compared to the low lands (Sun *et al.,* 2017).

ET changes also have an effect on streamflow response (Gallart *et al.*, 2002). Jiang *et al.* (2012) revealed a significant negative correlation between streamflow and NDVI. The growing season is associated with increased vegetation activity, temperatures and the resultant ET. The negative correlation between streamflow and NDVI is underpinned by vegetation which promotes infiltration and groundwater recharge. This interception of surface runoff by vegetation has implications for reducing surface runoff-streamflow contributions. Hence, low streamflow values are measured in catchments with high vegetative cover (Jiang *et al.*, 2012; Wang *et al.*, 2012a).

However, the degree at which vegetation affects streamflow is also dependent on the amount of rainfall. High streamflow values can be measured where rainfall amount exceeds the soil infiltration and water retention capacity (Odiyo *et al.*, 2015; Akpoti *et al.*, 2016; Martins *et al.*, 2016). Nevertheless, if rainfall is below the catchment infiltration and water storage capacity then low streamflow prevails. Thus, it can be inferred that vegetation has direct effects on streamflow response. Therefore, ET changes can be reflective of vegetation effects on streamflow response.

The negative correlation between streamflow and ET could suggest profound impacts of ET on soil moisture loss which then reduces surface runoff and contributions to streamflow owing to enhanced soil hydraulic conductivity (Akpoti *et al.*, 2016; Gyamfi *et al.*, 2016). Zhang *et al.* (1999) emphasised that changes in annual surface runoff attributed to land use change such as vegetation cover change should be reflected in annual ET.

Furthermore, Sun *et al.* (2017) observed a significant increase in annual ET because of shrubland and grassland succession in degraded sections of a catchment. Degraded catchment landscapes tend to be dysfunctional and support little plant activity. Thus, low ET can be measured owing to minimal or lack of plant activity. Therefore, the increase in annual ET can be attributed to reestablishment of vegetation in the previously degraded parts of a catchment.

2.2.3 Grassland encroachment by shrubs: Implications for catchment hydrological processes

Native vegetation structure, biomass and spatial distribution patterns are greatly modified by invasions and in turn catchment hydrological response is affected (Brooks, 2003; Jayiya *et al.*, 2004). Woody shrub invasion in catchments has adverse impacts on soil surface conditions, resident vegetation degradation as well as alteration of catchment hydrologic response. According to Kakembo (2004; 2009), rangeland degradation is driven by *Pteronia incana,* a patchy annual dwarf shrub associated with soil surface crusting. Continual degradation of the native vegetation cover and expansion of crusted inter-patch soil surface areas facilitate generation of surface runoff and subsequent soil erosion, owing to impeded infiltration.

The encroachment of a catchment by the invader shrubs induces conversion of catchment hillslopes to dysfunctional landscapes, characterised by loss of soil, water and nutrients (Kakembo, 2004; Kakembo *et al.*, 2007; Odindi and Kakembo, 2011; Manjoro *et al.*, 2012a). Studies on catchment encroachment by *P. incana* have not looked at invasion impacts on catchment hydrological responses. Nonetheless, literature suggests the existence of the link between encroachment and catchment hydrologic response. Reduction of native vegetation, soil surface crusting and soil erosion linked to shrub invasion are indicative of altered catchment hydrologic response (Odindi and Kakembo, 2011; Kakembo *et al.,* 2012). Odindi and Kakembo (2011) noted that the invader reduces soil moisture content in invaded hillslopes.

The study highlighted that *P. incana* invasion has implications for surface runoff and landscape functionality.

2.2.4 Invasive Alien Plants' (IAPs) impacts on catchment hydrological processes

Amongst other contributing factors to the disruption of hydrological responses and land degradation are Invasive Alien Plants (IAPs). Dzikiti *et al.* (2013) highlighted that IAPs have great impacts on streamflow reduction, lowering of groundwater reserves, reduction of grazing land, loss of biological diversity, exacerbation of dramatic wild fires and severe soil erosion. This chain of impacts by IAPs has serious implications for the ecohydrological integrity of catchment areas.

The impacts of IAPs on streamflow reduction have been quantified and demonstrated by several studies (Van Wyk, 1986; Bosch 1979; Le Maitre *et al*., 1996; Rwizi, 2015; Le Maitre *et al.*, 2015). Bosch (1979) recorded an 82 % reduction of streamflow over a period of 20 years in the Drakensberg, KwaZulu Natal, following plantation of pine trees in a grassland catchment. Van Wyk (1986) quantified a streamflow reduction of 55 % after invasion of fynbos by pines. The findings by Bosch (1979) and Van Wyk (1986) are in line with the observation made by Le Maitre *et al.* (2015). According to Le Maitre *et al.* (2015), the impacts of IAPS are more pronounced where there is a great difference in vegetation structure (e.g. Leaf Area Index (LAI), Height, and roots depths). Bosch (1979) and Van Wyk (1986) highlighted a reduction in streamflow attributed to the invasion of pine trees in grassland and shrubland. Pines have greater vegetation traits and water use requirements compared to grass and shrubs. Hence, the impacts of pine invasions tend to be more pronounced on water flow (Bosch, 1979; Dye and Poulter, 1995; Dzikiti *et al.*, 2013).

The reduction of streamflow by IAPs has been attributed to their high-water consumption associated with high transpiration, deep and widely extending rooting systems that maximise water use over native species (Calder and Dye, 2001). Moyo *et al.* (2009) state that *Acacia Mearnsii* (Black Wattle) significantly modifies hydrological responses through the high evapotranspiration rates that facilitate water loss and lowered water tables.

A similar study by Gorgens and Wilgen (2004) found that both shrubs and tree IAPs decrease surface runoff and groundwater recharge through increased above-ground biomass coupled by high evapotranspiration. Le Maitre and Versefeld (2000) reported an estimate of 3 300 Mm^3 surface runoff reduction by IAPs, which is 7% of the national total, was quantified mostly from fynbos shrubland and grassland catchments. This great reduction of surface runoff could be indicative of blanket invasions of tree invaders within the respective catchments.

More recently, Le Maitre *et al.* (2015) noted that in dryland catchment areas, limited rainfall and water availability for plants tend to be the major factors regulating evapotranspiration and surface runoff. An estimated surface runoff increase of 300-400 mm/year is said to be induced where indigenous vegetation is seasonal grassland invaded by evergreen IAPs. Where indigenous vegetation is evergreen shrub species, surface runoff increases are approximately 200-300 mm/year (Le Maitre *et al.*, 2015). These findings highlight the effects of vegetation structure difference and effect of seasonality on influencing the impacts of IAPs. Seasonal native grassland invasion by evergreen IAPs generates more surface runoff during the non-growing season because of predominantly bare catchment surfaces. Thus, more surface runoff is generated from catchments with seasonal native grassland compared to the catchments with evergreen shrubs which afford vegetative cover throughout the year to minimise generation of greater surface runoff. Cullis *et al.* (2007) reported that the reduced annual mean runoff by IAPs consequently reduces water yield in catchment areas.

Le Maitre *et al.* (2015) further observe that the impacts of invaders are also more pronounced in areas of high water availability such as shallow table zones of the topography and riparian settings. The authors reveal that water use by invaders can increase 1.5 to 2 times compared to the same native species, particularly in dryland catchment areas. They further point out that invaders have greater water use impacts in riparian settings compared to dryland invasions.

Conversely, Calder and Dye (2001) conclude that in dry climatic regions the impacts of invaders in water use are more pronounced in water limited areas rather than in water unlimited areas such as riparian zones. The different conclusions on the impacts of invaders on water use in dry climates imply that impacts of IAPs on water use are area-specific and may be determined by several other factors. Cullis *et al.* (2007) also argue that the impacts of IAPs differ for different states of invasion.

Catchment vulnerability to severe soil erosion is also attributed to the occurrence of intense fires favoured by IAPs**.** Fire events resulting in burning of invaders cause water-repellent soil layers which are attributed to enhanced soil erosion during and after rains (Smith *et al*., 2011). Soil hydrophobicity is a mechanism introduced post fires (Adams *et al.*, 1970). The mechanism is produced by the presence of water-repellent materials such as resins which are volatised in the burning process and then distil and descend according to the existing soil thermal gradient within the soil profile (Adams *et al.*, 1970; Scott and Van Wyk, 1990; Smith *et al.*, 2011).

Adams *et al.* (1970) recorded that 50-59% of substances produced by IAPs during fire to be potential inducers of water repellence in soil. As a result of the mechanism of soil hydrophobicity, water infiltration, groundwater recharge and/or percolation are reduced. Consequently, surface runoff increases and enhances soil erosion (Scott and Van Wyk, 1990). Increased sediment injection to watercourses is induced by high soil erosion caused by soil hydrophobicity.

Smith *et al.* (2011) reported that transport of sediments to streams tends to be enhanced up to 1 459 times post fire events. Further, Smith *et al.* (2011) added that slope is a critical determinant of sediment loads post fire events.
Smith and Scott (1992) reported that 15° slope angle produced ca. 9.9 t/ha per year of sediments whilst a 32^{o} slope yielded 25.9 t/ha per year of sediments. Therefore, the quantity of sediment yield is a function of slope angle. The injection of sediments to streams degrades water quality and quantity. The modification of fire regimes perpetuates soil erosion and the associated sediment yield, which in turn affects water quality (Chamier *et al.*, 2012).

Other studies have highlighted the impacts of invasive plants on water quality (Nagler *et al.*, 2008; Chamier *et al*., 2012). Nagler *et al.* (2008) quantified the impacts of *Tamarix chinensis* on groundwater quality and their findings showed that groundwater salinity quadrupled from 225 mg/l to 10 000 mg/l owing to elevated evapotranspiration in the Colorado River system, in the US. Chamier *et al.* (2012) state that reduced streamflow and groundwater recharge induced by high evapotranspiration rates yield less riverine water dilution and high concentrations of nutrients and suspended solids, respectively. These degrade riverine water quality. Furthermore, the increase in IAPs' biomass goes hand in hand with litter production, and this is said to induce potential alterations in physico-chemical properties of the soil which could potentially affect water quality (Chamier *et al*., 2012).

The impacts of IAPs are discussed here due to the fact that invaders form part of the catchment land cover and are major drivers of land cover changes and the subsequent catchment hydrological response. Therefore, understanding their role and impacts is essential in the quest to gain a comprehensive understanding of the hydrologic impacts of land use/cover change.

2.2.5 Overgrazing impacts on hydrological responses and water quality

Overgrazing is often attributed to degradation of grassland cover and ecosystems in arid and semi-arid regions' catchments (Kashaigili and Majaliwa, 2013; Li *et al.*, 2016; Sun *et al.*, 2017). Precipitation interception, infiltration and groundwater recharge, surface runoff and streamflow and evapotranspiration are some hydrological processes that are affected by vegetation degradation caused by overgrazing. Finca *et al.* (2011) state that poor land stewardship associated malpractices such as overgrazing and mistimed veld fires also greatly affect water resources.

Generally, overgrazing coupled by animal trampling, induce soil surface crusting and sealing which has been explained to impede infiltration and exacerbate surface runoff generation. Overgrazing leads to critically low vegetation cover, poor grassland health, landscape dysfunctionality and accelerated land degradation via severe soil erosion (Finca *et al.*, 2011). Excessive sediment yield resulting from soil erosion is a major contributor to stream aggradation, high turbidity levels and siltation of dams (Baird and Heymans, 1996; Rowntree and Wadeson, 1998; Odiyo *et al.*, 2015). These processes deteriorate water quality and reduce water quantity. Dam siltation gradually reduces dams' depth and volume, thus exposing dams to high rates of evaporation and leading to water loss. In addition, catchment areas prone to overgrazing and animal trampling are characterised by increased and reduced streamflow during the rainy and dry season respectively (Sun *et al.*, 2017a).

2.2.6 Hydrologic impacts of catchment impoundments

"Fluvial systems are sensitive to human impacts such as injudicious land use and other direct human impacts such as channel impoundments" (Mhangara *et al.*, 2011; Rowntree and Dollar,1994). According to Mhangara *et al.* (2011), Rowntree and Dollar (1994) observe that the construction of the Sandile Dam in 1981 had severe impacts on the natural functioning of the Keiskamma river. Rowntree and Dollar (1994) state that the introduction of the dam to the

Keiskamma hydrologic system led to a reduction of sediment transportation capacity, channel siltation as well as a cascade of ecological problems.

High flow regulation and 100% sediment trap efficiency were amongst the impacts associated with the impoundment (Rowntree and Dollar, 2008). Most of the high discharges are absorbed by the impoundment and downstream flow regimes were reduced (Rowntree and Dollar, 2008). Downstream reduction of high flows and elimination of infrequent greater magnitude flows has also contributed to channel sedimentation and aggradation owing to scarce, if not eliminated, periodic flushing of the fluvial system (Hickin, 1984). Since the impoundment in 1981, flood frequency curves showed a reduction of up to 30% in peak flows (Rowntree and Dollar, 1984).

In 1993, following a single rainstorm event, a discharge of 2.46 $m^3. s^{-1}$ was quantified upstream Keiskamma river whilst 0.1 $m^3.s^{-1}$ was measured downstream (Rowntree and Dollar,1994). This finding clearly shows the impacts of channel impoundment on flow regulation, in this case by a 95.9 % streamflow reduction. Downstream loss of saturation is one of the hydrological responses of river impoundments (Stromberg *et al.*, 1996; DeWine and Cooper, 2007; González *et al.*, 2010). These impacts affect downstream vegetation water requirements including both riparian and bordering hillslope vegetation. The prevalence of such conditions gradually creates a niche for invasive plant species to thrive through outcompeting resident vegetation on the water deficit environments (Baird and Heymans, 1996; Rowntree and Wadeson, 1998).

2.3 Hydrological modelling

This section of the literature review elucidates the process of hydrological modelling, classification and types of hydrologic models. First, background on hydrological modelling is given, covering factors that have influenced development of the models. General characteristics and input data for the models are explained as well. Secondly, hydrological model classification is made and a review of selected hydrological models suitable for arid-semi arid environments is done. The description of the selected models focuses on the general description of the model, input data, output processes (simulations), advantages and limitations.

Finally, this section reviews the model selected for the study, the Soil and Water Assessment Tool (SWAT) hydrological model and motivates the choice of the model for this study.

2.3.1 Overview of hydrological modelling

Prior to the birth of hydrological modelling, field experiments were the traditional methods employed in investigating hydrological responses arising from land use/cover changes and/or vegetation changes. One shortcoming of field experiments is the limitation of capturing a few factors that may be limited to field or hillslope scale and which may not significantly affect the catchment's hydrological response (Devia *et al.,* 2015). Moreover, catchment hydrological response is dependent on many factors such as vegetation spatial patterns, soil types and the associated physical and chemical characteristics, groundwater and rainfall trends (Zhang *et al*., 1999). To study all the factors in large catchments has proved to be expensive and difficult when using field experiments. Yet, knowledge of the connections between the factors of vegetation-hydrological responses remains of fundamental important for catchment management. To overcome the limitations associated with field experiments, hydrological models were developed. Nonetheless, hydrological models differ and are also not entirely immune to bias (Gyamfi *et al.*, 2016) . More details on hydrologic models are reviewed in the subsequent sub-sections.

Hydrological modelling has evolved over time and this has been influenced by changes in the biophysical environment. Phenomena such as land cover/use changes, climate change, and heterogeneity in soils have shaped the hydrological modelling environment. The changes in the natural environment have direct impacts on hydrological elements such as surface runoff and discharge. Thus, hydrological models ought to comprehensively account for critical factors that affect the processes of interest. Devia *et al.* (2015) state that each hydrological model has its inherent unique attributes. The input datasets used by hydrological models include climatic data, rainfall, air temperature, humidity, wind speed, solar radiation, topography, soil characteristics, vegetation and several other physical parameters (Gan *et al.*, 1997; Gupta *et al.*, 1999; Devia *et al.*, 2015).

2.3.2 Classification of hydrological models

Hydrological models are simplified representations of the real world hydrological system and its processes. Hydrological models are applied in predicting and understanding hydrological processes. Nowadays hydrologic models are integral tools for water resource management.

The classification of models, particularly rainfall-runoff models, is founded on the model's input datasets and parameters and physical principles governing application of the model (Xu, 2002; Akpoti *et al.*, 2016). Models can be classified into lumped and distributed types based on parameters of the model as a function of spatiality and temporality and the model's nature being deterministic and stochastic (Xu, 2002).

The common feature about deterministic and stochastic models is that they use mathematical equations with parameters and variables as part of the model. Deterministic models use known or assumed parameters. The model produces its results by means of statistical methods like linear-regression and or non-linear curve procedures (Xu, 2002).

Deterministic models have reduced uncertainty and comprise better understood heterogeneity compared to stochastic model. However, the two differ in that a stochastic model considers the presence of some randomness in one or more of its parameters. The results of stochastic models are conventionally expressed in probabilistic terms, as the presence of randomness usually yields results with errors and some uncertainty. Devia *et al.* (2015) state that in deterministic models, a single set of values outputs the same results, whilst for stochastic models, different output values are produced from a single input dataset.

On the basis of the work by Moradkhani and Sorooshian (2008), Devia *et al.* (2015) described lumped models as inherently considering a catchment area as a single and homogenous unit with no spatial variability and outputs are generated without considering a catchment's spatial heterogeneity. Conversely, distributed models' predictions consider a catchment's spatial variability and a catchment is divided into small units with variable parameters, inputs and outputs. In reality, catchments are environments that characteristically have diverse conditions and therefore, distributed models are more realistic than the lumped counterparts (Xu, 2002).

Models can also be classified into static and dynamic models based on time. Dynamic models emphasise temporal changes of spatial data and interactions between variables whilst static models exclude time scales (Xu, 2002). Furthermore, models can be classified as continuous and event based. The latter outputs strictly for specific time periods, whilst the former simulates continuous output. Conceptual models, empirical models and physically based models form the core of hydrological models' classification (Nash and Sutcliffe, 1970; Gan *et al.*, 1997; Xu, 2002; Devia *et al.*, 2015).

Empirical hydrological models

Empirical hydrological models are based on observations and derive their information from existing data. The modelling process of these models does not consider the catchment's hydrological system and features. Devia *et al.* (2015) and Xu (2002) described them as data driven models. Hydrographs are an example of these types of models. These models use coexisting input and output time series to derive their mathematical equations.

Finally, regression and correlations models are adopted by empirical models to test for functional relationships between input and output variables (Legates and McCabe, 1999; Adeogun *et al.*, 2014; Devia *et al.*, 2015; Woldesenbet *et al.*, 2017).

Conceptual hydrological models

In conceptual models, all components of the hydrological processes are described. The models inherently consider several interlinked basins representing the physical units of a catchment (Nash and Sutcliffe, 1970; Arnold *et al.*, 2012b; Devia *et al.*, 2015). The basins are described as subject to recharge processes through rainfall, infiltration and percolation as well as water loss processes such as runoff, drainage and evapotranspiration (Legates and McCabe, 1999). These models use field data and model calibration techniques to assess model parameters. In addition, semi empirical equations are employed in these models. Devia *et al.* (2015) point out that the calibration process makes interpretation convenient because of the curve fitting involved. Thus, land use change effects cannot be projected with firm certainty based on the reliability of the predictions (Devia *et al*., 2015).

Physically based models

Physically oriented models are founded on mathematical representations of the real world hydrological phenomena (Jakeman and Hornberger, 1993; Devia *et al.*, 2015). These models include the fundamental principles of physical processes; hence they are also referred to as mechanistic models. The models use measurable variables which are functions of both spatial and temporal scales. Infinite difference equations are used to represent movement of hydrological processes (Xu, 2002). Calibration of these models is free of extensive meteorological and hydrological data requirements. Nonetheless, physical characteristics of a catchment need to be described through the evaluation of several parameters. The models initially require a lot of input data such as meteorological, hydrological, soils, topography and basin physiography. Physically based models can be applied at various ranges of scenarios.

Table 2.1.Classification of hydrological models

Source: (Devia *et al*., 2015: p1003).

Empirical model	**Conceptual model**	**Physically based model**
Data based or metric or black box model	Parametric or grey box model	Mechanistic or white box model
Involve mathematical equations, derive value from available time series	Based on modelling of reservoirs and include semi empirical equations with a physical basis	Based on spatial distribution. Evaluation of parameters describing physical characteristics
High predictive power, low explanatory depth	Simple and can be easily implemented in computer code	Complex model. Requires human expertise and computation capability
Cannot be cannot be extrapolated to other catchments	Require extensive hydrological and meteorological data	Suffer from scale related problems
ANN, unit hydrograph	HVB model, TOPMODEL	SHE, or MIKESHE model, SWAT
Valid within the boundary of given domain	Calibration involves curve fitting making physical interpretation difficult	Valid for a wide range of situations

2.3.3 Hydrological models selected for review

Variable Infiltration Capacity (VIC) model

The VIC model is also a semi distributed hydrological model (Devia *et al.*, 2015; Hamman *et al*., 2018). The model is grid based and uses energy and water balance equations to model hydrological phenomena. Major input data include precipitation, minimum and maximum daily temperature, wind speed and land cover types encompassed by the model grid. Empirical equations govern processes such as infiltration, surface runoff and saturation flow. Infiltration excess runoff and saturation excess runoff generate surface runoff. Saturation excess runoff simulation considers precipitation and soil heterogeneity (Maurer, 2011; Devia *et al.*, 2015; Hamman *et al.*, 2018).

The model consists of three embedded layers that modulate hydrological behaviour within a catchment. The first, top layer facilitates soil evaporation, middle layers describe dynamics of soil response to precipitation and the bottom layer characterises soil moisture behaviour. According to Devia *et al.* (2015), the up-to-date VIC model for surface runoff considers infiltration and saturation excess runoff as well as the influence of heterogeneity in soils. The model is nowadays applied to several catchments to investigate the hydrological impacts of climate and land cover changes.

The Hydrologic Engineering Centre-Hydrologic Modelling System (HEC-HMS)

HEC-HMS is a highly integrated hydrological model that is designed for simulation of hydrological processes in catchments with a dendritic drainage pattern. Traditional hydrological analysis procedures are embedded within the model. These procedures include unit hydrographs, hydrological routing and infiltration. The model includes capabilities to simulate processes such as evapotranspiration, snowmelt and soil moisture regimes over long temporal scales (Fleming, 2010). Additional analysis tools for optimisation of the model, streamflow predictions, erosion and sediment yield water quality form part of the model's capabilities. In addition, the model can also perform gridded runoff simulations employing a method knowns as the linear quasi-distributed runoff transform (Scharffenberg, 2016).

The HEC-HMS has four (4) components; namely (1) basin model, (2) meteorological model, (3) control specifications, and (4) input data. The model simulates rainfall runoff response in the basin model utilising inputs from the meteorological model. The basin model component represents the physical catchment (Fleming, 2010; Scharffenberg, 2016). The basin model is developed by integrating certain hydrological elements which physically define the catchment using mathematical equations. Hydrological elements include sub-basins, reaches, junctions, sources, sinks and reservoirs. The meteorological model component functions to compute precipitation input as per sub-basin element requirements. Point and gridded precipitation can both be used by the model. The control specification component of the system governs timeframes for simulations. Input data components of the system are divided into three categories, *viz*: time-series data, paired data and gridded data (Scharffenberg, 2016).

Hydrographs produced by the programme can be used directly or in conjunction with other software for studies of water availability, urban drainage, flow forecasting, future urbanisation impacts, reservoir spillway design, flood damage reduction, floodplain regulation, wetlands hydrological responses and systems operations (Scharffenberg, 2016).

IHACRES

IHACRES is an abbreviation for **I**dentification of unit **H**ydrographs **a**nd **C**omponent flows from **R**ainfall, **E**vaporation and **S**treamflow data. IHACRES is a parametric conceptual rainfall-runoff model. It has been applied successfully in several arid and semi-arid catchment areas (Croke and Jakeman, 2004, 2008; Croke *et al.*, 2004). The model was developed by Jakeman and Hornberger (1993)and Jakeman *et al.* (1990). The model is a catchment-scale modelling tool. It models the rainfall-streamflow dynamics from small catchments to large watersheds at various temporal scales.

Croke and Jakeman (2004) state that IHACRES typically requires between 5 and 7 parameters. The model requires rainfall (mm) and temperature (^{o}) or evapotranspiration (mm) data to simulate catchment streamflow. It is a lumped model because it does not require any geospatial data such as DEM, soils and land use maps. Initially, the model must be calibrated over a known period to optimise the model's performance. In ungauged catchments, the model simulates streamflow. Other studies have used the model to investigate the hydrologic response of land use/cover change (Croke and Jakeman, 2004, 2008; Croke *et al.*, 2004).

2.4 Selected hydrological model for the present study

The SWAT hydrological model is the selected model for this investigation.

2.4.1 Soil and Water Assessment Tool (SWAT) hydrological model

The SWAT model is a physically based, semi-distributed and process based hydrological model. SWAT operates using daily time steps; it is developed and designed to predict land use and management impacts on water (Arnold *et al.*, 2012b). The core components of the model are weather, hydrological responses, soils, plant growth, nutrients, pesticides, bacteria and pathogens and land management (Arnold *et al.*, 2012a).

Major elements such as nutrients, pesticides, bacteria and pathogens are of utmost importance for agricultural water contamination investigations. For purposes of this study, such elements are not important. Weather, hydrological responses and soils are considered relevant.

The model divides a catchment area into several sub-catchments which are further subdivided into Hydrologic Response Units (HRUs). HRUs are characterised by homogeneity in topography, soil properties, land use and management. HRUs are neither contiguous not spatially recognised in simulations. They are technically represented as a percentage of the sub-catchments. In some cases, catchment division gets limited to sub-catchment level, should there be no further heterogeneity in catchment characteristics necessitating delineating up to HRUs (Arnold *et al.*, 2012b).

The SWAT model requires four main data inputs files (1) Digital Elevation Model (DEM), (2) meteorological data, (3) soils data and (4) land use data. A DEM contains catchment topographic characteristics. The required meteorological data include daily precipitation, maximum and minimum air temperature, solar radiation, wind speed and relative humidity. Soil data required by SWAT can be divided into mandatory physical characteristics and optional chemical characteristics. Land use data can be in the form of maps or management schedules. There are many other input datasets for SWAT but they are optional (Arnold *et al.*, 2012b).

All processes in SWAT are fundamentally driven by the water balance because it has impacts on model elements such as the plant growth module and sediment movement (Arnold *et al*., 2012b). The model partitions its simulation process into two phases, *viz*; land phase and routing phase. The land phase is responsible for controlling the amount of material entering the main channel per sub-basin from hillslopes (Arnold *et al*., 2012b). The movement of water, sediment and associated material through the channel to the catchment outlet is controlled by the routing phase. These two phases of the model are representative of real world hydrological pathways. Basically, hillslope water pathways feed into the main channel and then fluvial processes occur up to the mouth of a channel (Neitsch *et al.*, 2002; USDA, 2004; Shekhar and Xiong, 2008).

Hydrological processes simulated by SWAT include canopy storage, surface runoff, infiltration, evapotranspiration, lateral flow, tile drainage, redistribution of water within the soil profile, consumptive use through pumping (if any), return flow, and recharge by seepage from surface water bodies, ponds, and tributary channels (Arnold *et al*., 2012). The model has been applied to simulate hydrological processes over long temporal scales (Githui *et al.*, 2009; Dos Santos *et al.*, 2014; Perry, 2014; Kalcic *et al.*, 2015; Guo *et al.*, 2016).

2.4.2 Calibration and validation of SWAT

Sensitivity analysis

Prior to model calibration, sensitivity analysis is the first important step to be carried out by the modeller. Sensitivity analysis is defined as the process where the rate of change in the model's output with respect to change in the model's input parameters is determined (Moriasi *et al.*, 2007; Arnold *et al.*, 2012b). Key parameters and parameter precision are required for the process of calibration (Moriasi *et al.*, 2007; Arnold *et al.*, 2012b). According to Moriasi *et al.* (2007), two types of sensitivity analysis; *viz* local and global, can be performed for a modelling project. Local sensitivity analysis is where values are adjusted one at a time. Conversely, the global method simultaneously allows all values to change (Moriasi *et al.*, 2007). Nevertheless, the two models also have their cons. According to Moriasi *et al.* (2007), the disadvantage of local sensitivity is that the correct values of fixed parameters are never known. The large number of simulations required in global sensitivity analysis render it time consuming as a limitation (Moriasi *et al.*, 2007).

Model calibration and validation

Model calibration is defined as the process of estimation of model parameters through comparing model output with observed data for the same period, whilst simultaneously adjusting parameters to achieve a good performance rating (Moriasi *et al*., 2007). According to Arnold *et al.* (2012b), model calibration is a critical effort to optimise model parameters to a set of local conditions, reducing uncertainty in the model's simulation.

Gan *et al.* (1997) state that validation and evaluation results are directly affected by the data used for calibrating the model's simulation. It is recommendable that data from a period of 3-5 years should be used for calibration, so that they are sufficiently comprehensive of ranges of hydrological events for activating all model processes at calibration (Neitsch *et al.*, 2002). Model simulations tend to demonstrate poor performance for shorter time periods compared to longer periods (Moriasi *et al*., 2007). Arnold *et al.* (2012b) and Moriasi *et al.* (2007) strongly recommend that baseflow and surface runoff separation from total streamflow should be done for both the calibration and validation period. The baseflow filter which was developed by Arnold *et al.* (1995) and later modified by Arnold and Allen (1999), can be used to execute this requirement.

The final step in modelling is validation which is the process of demonstrating that a model can calibrate beyond novel conditions and environments. Arnold *et al.* (2012b) recommend that good model calibration and validation should encompass the following: first, observed data inclusive of all year conditions such as dry, wet and average; second, multiple model performance evaluation statistics; third, calibration for all aspects of interest and fourth, model's output verifications for reasonability.

2.4.3 SWAT model performance assessment methods

Model performance evaluation is one of the critical aspects of modelling. This important step is done concurrently with calibration and validation. Assessment of model performance differs for the calibration and validation phases of modelling (Gassman *et al.*, 2007). During calibration, model performance evaluation is an iterative process. Model parameters are adjusted until the desired performance value is achieved. For the validation phase, there is no adjustment of model parameters whatsoever. Thus, the initial model performance statistics produced for the simulation are reported (Arnold *et al.*, 2012b).

Conventionally, model performance evaluation is done using statistical and graphical methods. These methods ensure that the model's simulations are within a realistic range of uncertainty. "To use model outputs for tasks ranging from regulation to research, models should be scientifically sound, robust, and defensible" (Moriasi *et al*., 2007). Model performance evaluation is one of the very important modelling components that determine the models' fitness for use.

2.4.4 Statistical methods

Moriasi *et al.* (2007) categorised model evaluation techniques into three (3) categories; *viz* standard regression, dimensionless and error index. Each of the three categories entails several techniques. Nonetheless, for this study techniques per category shall be reviewed based on common application and suitability for this investigation.

Standard regression model evaluation statistics

According to Moriasi *et al.* (2007), standard regression statistical methods are for determining the degree of linearity of the relationship between simulated and observed data. The two widely and commonly used standard regression statics are (1) Pearson's correlation coefficient (r) and (2) Coefficient of determination (R^2). The correlation coefficient is an index of the degree of linearity between measured and simulated data, and it ranges from -1 to 1, where r = 0 indicates no existence of a linear relationship. If r = -1 and r = 1, this indicates a negative relationship and a perfect positive relationship, respectively (Legates and McCabe, 1999; Moriasi *et al.*, 2007; Adeogun *et al.*, 2014).

The coefficient of determination describes the proportion of the variance in observed data as explained by the model (Moriasi *et al*., 2007). The coefficient of determination ranges from 0 to 1, where high values indicate less error variance and vice versa. Despite the wide use of these two statistics, Legates and McCabe (1999) report on their oversensitivity to high extreme values and insensitivity to additive and proposal differences between simulated and observed data. Moreover, these issues highlighted by Legates and McCabe (1999) appear to be negligible, as most SWAT studies employ these standard regression evaluation statistics.

Dimensionless model evaluation techniques

A relative model evaluation assessment is provided by dimensionless techniques (Moriasi *et al.*, 2007; Arnold *et al.*, 2012b). There are many dimensionless model evaluation techniques; however, only the Nash-Sutcliffe Efficiency (NSE) will be reviewed for purposes of the present study.

NSE is defined as a normalised statistic used to determine the relative magnitude of the residual variance of simulations in comparison to the observed data variance (Nash and Sutcliffe, 1970; Moriasi *et al.*, 2007). NSE is an indicator of how well the plot of measured versus simulated data fits the 1:1 line. Mathematically, NSE is computed as shown in the equation below:

$$NSE = 1 - \left[\frac{\sum_{i=1}^{n}(Y_i^{obs} - Y_i^{sim})^2}{\sum_{i=1}^{n}(Y_i^{obs} - Y^{mean})^2}\right]$$

Where Y_i^{obs} is the i^{th} observation for the variable being evaluated, Y_i^{sim} is the i^{th} simulated value for the variable being evaluated, Y^{mean} is the mean of the observed data for the constituent being evaluated and n is the total number of observations (Moriasi *et al*., 2007, p 887).

NSE ranges between infinite (~) and 1.0 (1 inclusive). If NSE = 1, that is an optimal value. Values between 0.0 and 1.0 are generally viewed as acceptable levels of performance whereas values ≤ 0.0 indicate that the mean observed value is a better predictor than the simulated values, which indicates unacceptable performance (Moriasi *et al.*, 2007, p 887). NSE is highly recommended owing to the fact that it is very commonly applied and that provides modellers with extensive information on reported values (Moriasi *et al.*, 2007; Adeogun *et al.*, 2014; Sun *et al.*, 2017a).

Error Index model evaluation techniques

Error indices measure the deviations in the units of the data of interest (Legates and McCabe, 1999; Moriasi *et al*., 2007). Root Means Square Error (RMSE) and Percent bias (PBIAS) are the most commonly used error index statistics for model evaluation. Moriasi *et al.* (2007) indicate that a low RMSE value is considered optimal for model performance.

Gupta *et al.* (1999) explain that Percent bias (PBIAS) quantifies the mean tendency of the simulated data to be larger or smaller than the measured data. Moriasi *et al.* (2007) observe that a PBIAS of zero is an optimal value of the statistic. PBIAS values of low magnitude are indicative of accurate model predictions (Moriasi *et al.*, 2007; Arnold *et al.*, 2012b). Gupta *et al.* (1999) further note that model underestimation bias and overestimation bias are reflected by positive and negative values, respectively. PBIAS is mathematically computed with the equation below:

$$PBIAS = \left[\frac{\sum_{i=1}^{n}(Y_i^{obs} - Y_i^{sim}) * (100)}{\sum_{i=1}^{n}(Y_i^{obs})}\right]$$

Where PBIAS is the deviation of data being evaluated, expressed as a percentage (Moriasi *et al.*, 2007, p888). PBIAS is also recommended for evaluations of model performance because it is very commonly used.

2.4.5 Graphical methods

Graphical model evaluation techniques are visualisation tools for comparison of simulated and observed data (Jakeman *et al.*, 1990; Moriasi *et al.*, 2007; Scharffenberg, 2016). Hydrographs and percent exceedance probability curves are two employed techniques for visual evaluation of model performance. Moriasi *et al.* (2007) further point to box plots and bar graphs as other graphical techniques that can be used to examine seasonality variations and distribution of data.

Hydrographs and percent exceedance probability curves

A hydrograph is defined as a time series plot of simulated and observed flow during the calibration and validation period. Model bias, peak flows timing and magnitude differences and recession curves shape can be identified in a hydrograph (Moriasi *et al.*, 2007). Percentage exceedance probability curves often present daily flow duration curves (Crooks and Naden, 2007; Moriasi *et al.*, 2007). The curves can illustrate how well the frequency of observed daily flows data is simulated throughout the calibration and validation periods.

According to the United States Environmental Protection Agency (2002), should the model's simulation not produce acceptable results conforming to the general performance ratings, that may be indicative of five reasons. First, a significant difference existed between calibration and validation conditions. Second, there was inappropriate and inadequate model calibration. Thirdly, there was inaccurately measured data. Fourthly, the model required more detailed inputs than the ones computed. Lastly, the model was incapable to simulate the catchment processes of the area of interest. It should be noted that reasons for explaining the failure of any hydrological model to yield acceptable results is not only limited to these reasons. These are also general faults; model incompetency can be due to many other project-specific underpinnings.

2.4.6 General model evaluation ratings

The topic of model evaluation techniques does not provide directions on the acceptable ranges of values for the model performance statistics. To address the problem associated with no standardised conventional model performance rating, Moriasi *et al.* (2007) developed Table 2.2 using peer-reviewed publications to provide organised and standardised model performance statistics values. Noteworthy, performance ratings for RSR (RMSE) and NSE are for all parameters. However, PBIAS is parameter specific.

Table 2.2. Recommended statistics for general model performance ratings for a monthly time step simulation

Source: Moriasi *et al.* (2007)

Performance Rating	RSR	NSE	PBIAS (%)		
			Streamflow	Sediment	N, P
Very good	$0.00 \leq RSR \leq 0.50$	$0.75 \leq NSE \leq 1.00$	$PBIAS \leq \pm 10$	$PBIAS \leq \pm 15$	$PBIAS \leq \pm 25$
Good	$0.50 \leq RSR \leq 0.60$	$0.65 \leq NSE \leq 0.75$	$\pm 10 \leq PBIAS \leq \pm 15$	$\pm 15 \leq PBIAS \leq \pm 30$	$\pm 25 \leq PBIAS \leq \pm 40$
Satisfactory	$0.60 \leq RSR \leq 0.70$	$0.50 \leq NSE \leq 0.65$	$\pm 10 \leq PBIAS \leq \pm 25$	$\pm 30 \leq PBIAS \leq \pm 55$	$\pm 40 \leq PBIAS \leq \pm 70$
Unsatisfactory	$RSR > 0.70$	$NSE \leq 0.50$	$PBIAS \leq \pm 25$	$PBIAS \leq \pm 55$	$PBIAS \leq \pm 70$

It is highly and strictly recommended that performance ratings for the calibration period should be within 'Good" and "Very Good," owing to the fact that optimisation of model parameters is done only for model calibration, not validation (Xu, 2002; Moriasi *et al.*, 2007; Arnold *et al.*, 2012b; Morán-Tejeda *et al.*, 2015). Thus, model performance at calibration phase must be optimised.

2.5 Field methods for assessing hydrological responses

This section of the investigation reviews methods generally employed to assess hydrological responses in the field. The review encompasses field techniques used to investigate catchment surface runoff and soil moisture.

2.5.1 Surface runoff methods

Surface runoff can be determined using several direct and indirect methods. These include Gerlach troughs, microplots and macroplots, as well as runoff Curve Numbers.

Direct methods of quantifying surface runoff

Direct methods conventionally estimate runoff at plot scale. Direct runoff estimates are usually conducted concurrently with sediment yield. Collecting devices of different designs and sizes, commonly referred to as collectors, are mostly employed to quantify surface runoff. The direct methods include the famous Gerlach Troughs, plots, tanks and divisors.

Gerlach Troughs are named after their inventor, T. Gerlach. Gerlach Troughs are types of containers that are dug into the surface on a hillslope to catch overland flow and act as sediment traps (Romero-Díaz *et al.*, 1988). Gerlach Troughs consist of a collecting gutter physically let into the soil surface and connected to a small container on the downstream side (see Figure 2.1).

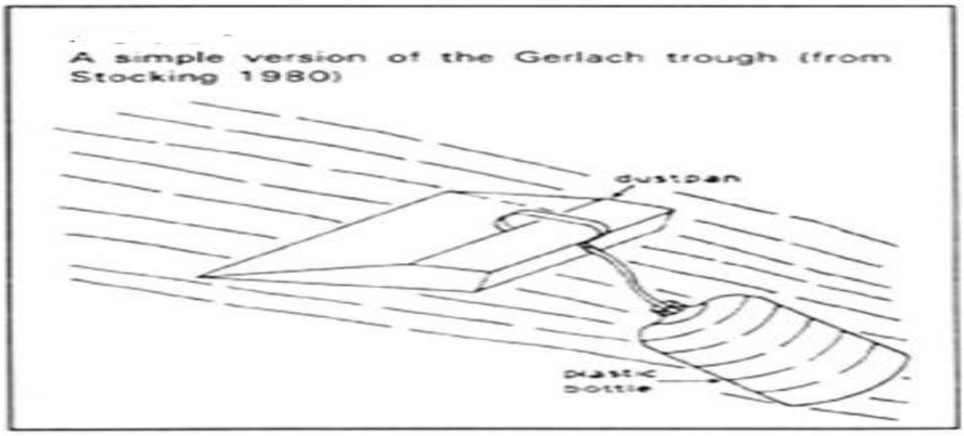

Figure 2.1. Gerlach Trough schematic illustration

Source: FAO (n.d)

Plots (Microplots) are also used to estimate surface runoff along hillslopes. Plots are of different sizes and designs. The Food and Agricultural Organisation (FAO) advises that designing of collectors must unarguably consider capacity of collectors to handle a maximal envisaged rate of flow and storage of probable maximum runoff quantity. Plots consist of two basic and fundamental common features; *viz*, a boundary and a collecting tank. The boundary channels overland flow to an outflow that empties into the tanks to collect the flow and transported sediments (see Figure 2.2).

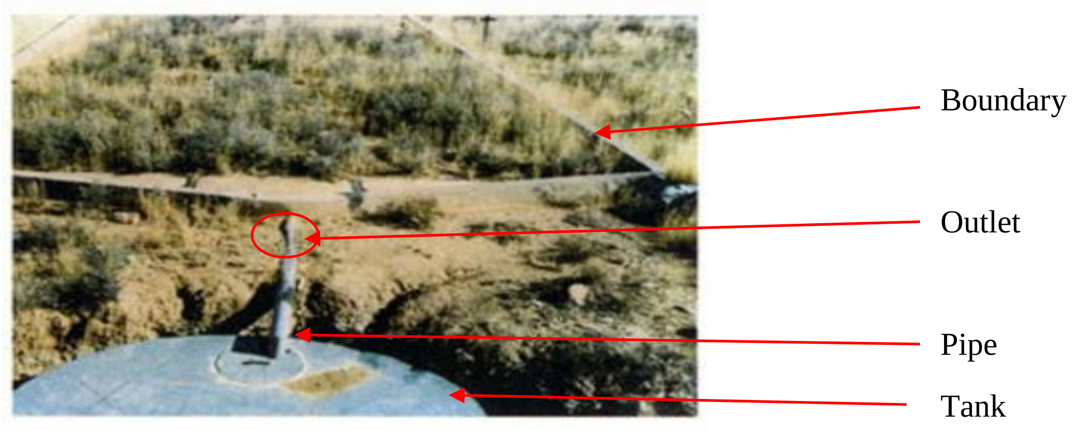

Figure 2.2. An example of a plot design

Source: FAO (n.d)

In larger plots where huge quantities of overland flow are captured, more than one tank is used to account and capture overflow. In such cases divisors are used to separate the flow and store it in different tanks (FAO, n.d) (see Figure 2.3).

Figure 2.3. Multiple pipe divisor

Source: FAO (n.d)

Quantification of surface runoff through employing direct methods is ideal for plot scale studies (Fox and Wilson, 2010; Canton *et al.*, 2011). Nonetheless, logistically the methods can be expensive to construct. Direct methods are also inherent of errors and poor estimations of surface runoff. Unforeseen extreme events are not entirely captured owing to inevitable overflows in collectors. Some of the methods are destructive to the environment. Construction of plot boundaries, and excavations for installation of tanks involve physical disturbances to the substrate (FAO, n.d). With regard to monitoring, direct methods require monitoring post extreme events and or over long temporal scales (Luppi *et al.*, 2009).

Figure 2.4. An example of G. Trough with minimised soil disturbance

Source: Rodrigo-Comino *et al*. (2017)

Indirect (surrogate) methods of quantifying surface runoff

Alternative to quantifying surface runoff on the field, standardised techniques can be employed to represent surface runoff conditions of different land cover types. The Runoff Curve Number (RCN) method was originally developed and established by the Soil Conservation Service (SCS) in 1954 (Bingner, 2000). The SCS Curve Numbers, or CN values, are used to estimate runoff from rainfall events as well as to be indicators of average runoff conditions. This is described to be essentially a coefficient that reduces total precipitation to runoff potential, after losses such as absorption, storage, transpiration and evaluation. Thus, a high CN value implies high potential runoff conditions. SCS runoff is derived from the following equation (NRCS, 1986).

$$Q = \frac{(P - I_a)^2}{(P - I_a) + S}$$

where

Q = runoff (in)
P = rainfall (in)
S = potential maximum retention after runoff begins (in) and
I_a = initial abstraction (in)

Further descriptions of the variables of the equation are detailed in NRCS (1986, p2).

"Determination of CN depends on the watershed's soil and cover conditions, which the model represents as hydrologic soil group, cover type, treatment, and hydrologic condition" (NRCS, 1986, p2). Table 2-2 (a to d) in NRCS (1986) contains a comprehensive database of CN values for determining average runoff conditions in urban, agricultural cultivated and related soil, as well as arid and semiarid areas.

A commonly highlighted limitation of the SCS CN values for runoff estimation is that they give average runoff conditions. Rainfall intensity and duration are not accounted for and accuracy of the method decreases when runoff is less than 12.7 mm (Bingner, 2000). Despite these limitations, the SCS runoff curve number methods have been used successfully in estimation of surface runoff conditions. The application of the method should consider all important considerations to produce robust and reliable runoff information.

This method can be applied to this investigation to demonstrate runoff conditions between grassland dominated catchments and shrub encroached catchments. Secondly, the SCS RCN method can be used to indirectly validate runoff simulations. The SWAT model also uses the SCS CN values to compute surface runoff simulations. Thus, model CN values can be compared to what is assigned manually to validate the model's surface runoff simulations.

2.5.2 Soil moisture assessments

Soil water content information is needed for a variety of other scientific investigations, such as climate change, environmental remediation, and engineering investigations (Yoram, 2003, page 3). Several techniques have been applied to quantify soil moisture.

Techniques used to measure soil moisture include but are not limited to the Gravimetric, Electrical Conductivity, Tensiometers and Radiometric methods (Johnson, 1962; Little *et al.*, 1998; Davidova and David, 2015). The Gravimetric method involves soil sampling and collection for weighing before and after drying the soil, where the difference in soil mass presents the actual soil moisture content (Johnson, 1962). The Gravimetric method is an ancient technique; however, it is still used owing to the fact that it is the only direct way to measure soil moisture. Limitations of the Gravimetric method are that it cannot measure soil moisture over a long temporal scale, it is labour intensive, requires impractical sample sizes for large areas and is unable to be conducted under certain weather conditions (Johnson, 1962; Little *et al.*, 1998). On the other hand, the Electrical Conductivity method uses soil moisture sensors which measure soil moisture content at specified depths over a long period. Tensiometers measure the matric potential of soil moisture in the field, making use of a porous clay cup attached to a tube filled with water (Johnson, 1962).

Davidova and David (2015) assessed the accuracy and applicability of different soil moisture measuring techniques. The Gravimetric method was the most accurate compared to electrical conductivity sensors (Davidova and David, 2015). Little *et al.* (1998) also investigated other proclaimed rapid soil measuring techniques as alternatives to the Gravimetric method. Their study indicates that the ThetaProbe and Speedy Moisture Tester are equally and significantly ($P<0.001$) as suitable as the Gravimetric method for undertaking soil moisture measurements. Findings by Davidova and David and Little *et al.* (1998) point to the consideration that different types of soil measuring techniques may perform differently under different conditions. Thus, it is important for researchers to be informed about the suitability and applicability of the various techniques (Little *et al.*, 1998; Walker *et al.*, 2004). According to Yoram (2003), radiometric and geophysical techniques are quick, accurate and reliable for measuring soil moisture content accounting for soil heterogeneity and considerable spatial extent.

Soil moisture measurement techniques are important for several research and management applications ranging from agriculture, land rehabilitation and water resource research. Moreover, applicability, suitability, accuracy and performance limitations should be considered for these techniques (Mittelbach *et al.*, 2011).

3 CHAPTER 3: STUDY AREA

Introduction

This chapter of the investigation presents the criteria for site selection of this study. The chapter further describes the site in terms of its geographic location, the catchment channel network, climate, vegetation, soils, geology, topography as well as demography and land use.

3.1 Site selection criteria

The Keiskamma tertiary catchment was selected to model hydrological impacts of land use/cover. The motive for applying the model at catchment scale was to capture environmental and anthropogenic factors that potentially influence the hydrologic response of the catchment. Catchment scale assessment of the hydrologic response to land use/cover changes is essential towards understanding the broad implications of land use/cover changes on the hydrology of the Keiskamma catchment (Legesse *et al.*, 2003; Le Maitre *et al.*, 2015). One of the prominent forms of land use/cover change in the catchment is shrub invasion. The shrub of interest was *Pteronia incana* (*P. incana*), which is a dwarf problem shrub that has invaded hillslopes of the Keiskamma catchment, particularly the dry central sections of the catchment. Thus, the investigation also sought to investigate hydrological impacts of the shrub encroachment on hillslopes affected by shrub invasion.

3.2 Description of Keiskamma catchment

3.2.1 Geographic location

The Keiskamma catchment (2442.93 km^2) is a semi-arid rural catchment located in the former Ciskei homeland of the Eastern Cape Province, South Africa (Mhangara *et al.*, 2011) (see Figure 3.1). The Keiskamma, which is the main river of the catchment, flows south westwards for 263 km and drains into the Indian Ocean at Hamburg estuary. The main tributaries of the catchment are Tyume, Gxulu and Chalumna. The word “Keiskamma” is derived from Khoisan people and it means sparkling rivers or waters (Haindongo, 2009). The catchment is located south of Hogsback and south west of Sutterheim in the Eastern Cape Province, South Africa. At the centre of the upper catchment section is Keiskammahoek town located at 32° 41′ 0″ S, 27° 9′ 0″ E.

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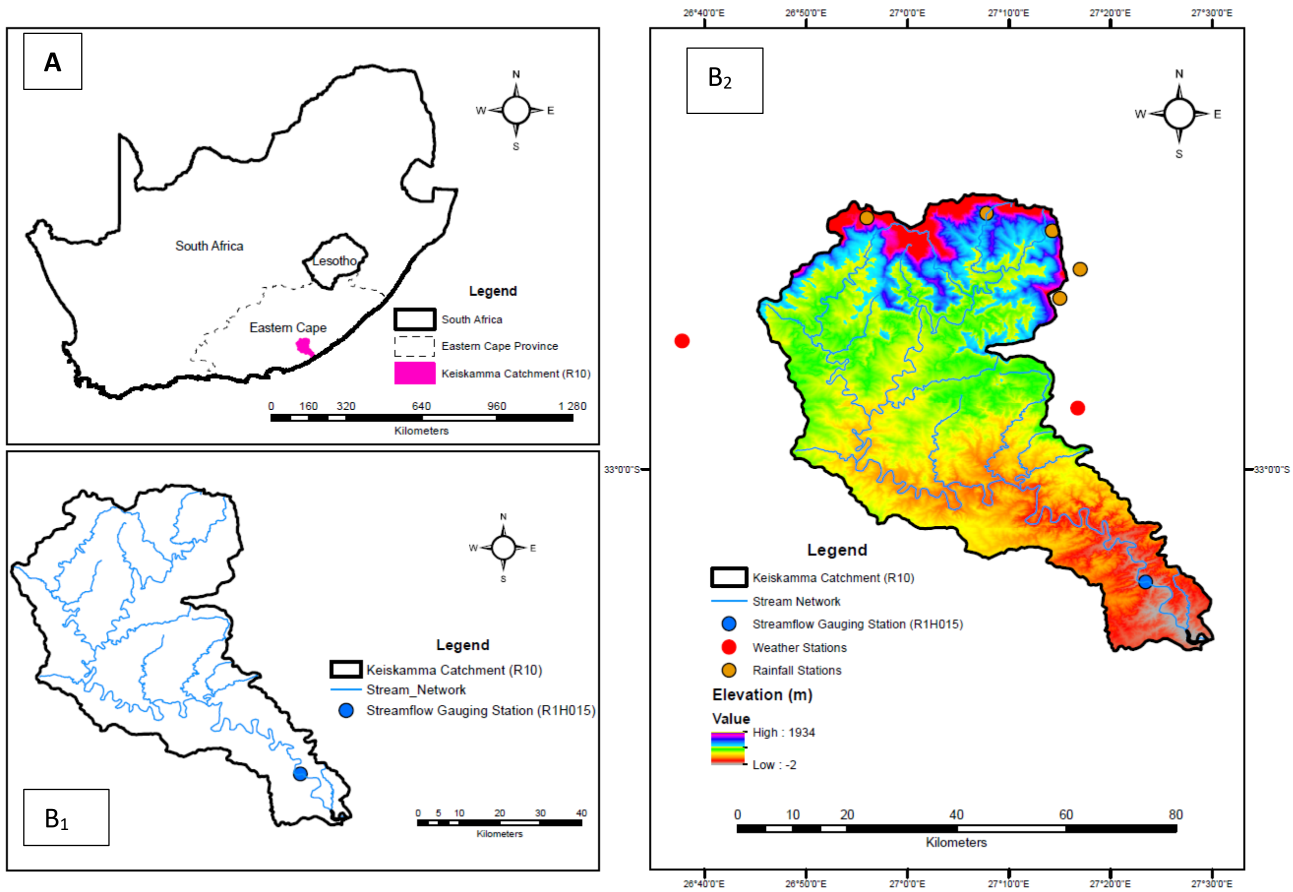

Figure 3.1. The catchment under study in the context of South Africa (A) and site specific map (catchment) (B_1 and B_2)

The catchment forms part of the Mzimvubu to Keiskamma Water Management Area. According to DWS catchment areas coding, Keiskamma is a Tertiary catchment represented as (R10) (DWS Government Gazette, 2016). The catchment has 4 quaternary catchments; *viz* R10A, R10B, R10 and R10D (DWS Government Gazette, 2016). The upper watershed of the catchment is the Amatole mountains. Amatole mountains are a Water Source Area of high importance. The mountains are one South Africa's High Water Yield Areas (see Figure 3.2).

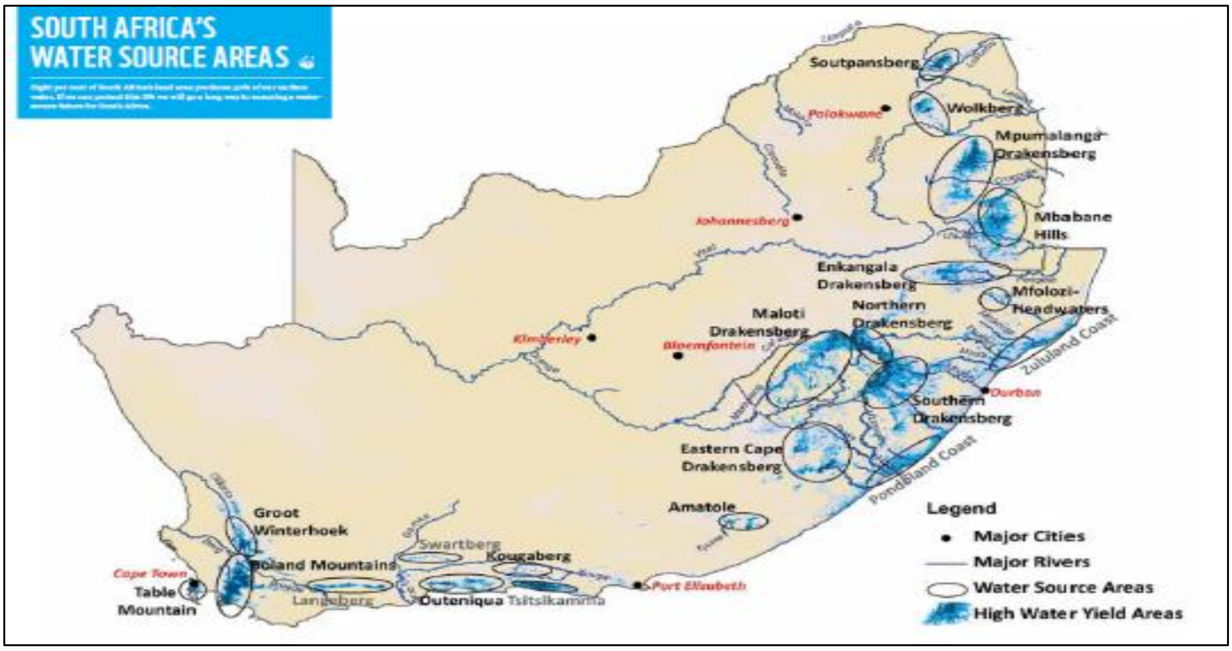

Figure 3.2. Map of South Africa's Water Source Areas

Source: WWF and CSIR (2013)

3.2.2 Catchment network

The upper watershed of the catchment is drained by three quaternary catchments; namely R10A, R10B and R10C which drain directly from the Amatole mountains. The three quaternary catchments have four major impoundments located within the upper parts of the catchment. The four impoundments are Binfield, Mnyameni, Cata and Sandile dams. Binfield dam is located ~11 km downslope of Hogsback in quaternary catchment R10C. Mnyameni dam is located in the upper reaches of quaternary catchment R10B. Cata dam is situated between Upper Gxulu (also known as KwaXhangwe) and Cata Village. Mnyameni dam supplies water to the Water Treatment Works (WTW) in Masincedane Location and Upper Mnyameni village (Amatola, 2015). The Mnyameni dam, in conjunction with the Cata dam, also supplies water for irrigation purposes to high lying agricultural lands within the Keiskamma valley (Amatola, 2015). The third major impoundment located on the transition between the upper and middle sections of the catchment is Sandile dam. Sandile dam provides raw water to the Sandile and Peddie Regional Water Treatment Works (WTW) for portable use (Amatola, 2015).

The upper watershed of the catchment is drained by six main annual rivers; namely Cata, Mnyama, Gxulu, Wolf, Keiskamma and Tyume Rivers. Cata River is about 8.49 km long from the headwaters to the Cata dam. Mnyama River spans 12.25 km from the upper watershed of the catchment to converge with Gxulu River at a point of confluence. Wolf river also feeds to Sandile from headwaters at Zingcuka location. Tyume river, which flows from the Hogsback mountains, drains the south-western side of the catchment. This tributary spans a length of 74 km from the headlands to meet the Keiskamma river north west of KuDikidikana area (-32.927231°; 26.960356°).

Two impoundments occupy the catchment's middle-lower reaches. These impoundments are Debe and Dimbaza dams. The middle to lower reaches of the catchments are drained by mostly ephemeral streams that join the Keiskamma River. The impoundments within the catchment significantly influence several fluvial processes and ecological dynamics.

3.2.3 Climate

Rainfall

The catchment is characterised by spatial and seasonal precipitation variability. Mhangara *et al.* (2011) attributed the climatic variations to elevation and its relative position to the sea. The mountain highlands Mean Annual Precipitation (MAP) ranges from 1500 to 1900 mm whilst it drastically drops to a range of 400 to 600 mm for the catchment coastal plateau. The catchment has predominantly summer rainfall and dry winters. According to DWAF (2002), winter seasons are not completely dry and tend to receive 30% of the annual rainfall which occurs between April and September. GIBB (2009) observes that for most Eastern Cape catchments, rain yielding systems include orographic forcing, frontal activity, convective mechanisms as well as tropical storms.

Temperature

The mean annual temperature for the escarpment zone is 11°C in winter. In summer, the temperature rises to 38°C, often exceeding 40°C, whilst in winter low temperatures of -2°C occur (DWAF, 2002; Mhangara *et al.*, 2011). Thus, upper reaches of the catchment on the Amatola mountains experience very cold winter months with very low temperatures and occasional snowfall events (Mbikwana and Bushula, 2008). The coastal plateau has annual average temperature of 18°C (Mbikwana and Bushula, 2008).

Humidity and evaporation

The relative humidity in the catchment is affected by seasonality. Relative humidity is higher in summer than in winter. According to DWAF (2002), during February relative humidity is generally highest with mean ranges from 60 to 82% and lower in July, ranging from 50 to 72%.

3.2.4 Vegetation

The catchment has a heterogeneous vegetation composition of both natural and exotic vegetation. This vegetation composition includes forest, thicket and grassland. The upper watersheds of the catchment are mountainous which are occupied by a dense evergreen forest and a dense valley thicket. The forest cover wanes with distance from the upper watershed of the catchment to assume a patchy distribution within which the forest stands, and dense bush is confined to river valleys. According to Rutherford *et al.* (2012), major vegetation units found on the upper catchment include the Southern Mistbelt Forest, Eastern Cape Escarpment Thicket, Buffels Thicket, Great Fish River Thicket and Amatole Montane Grassland (see Figure 3.3).

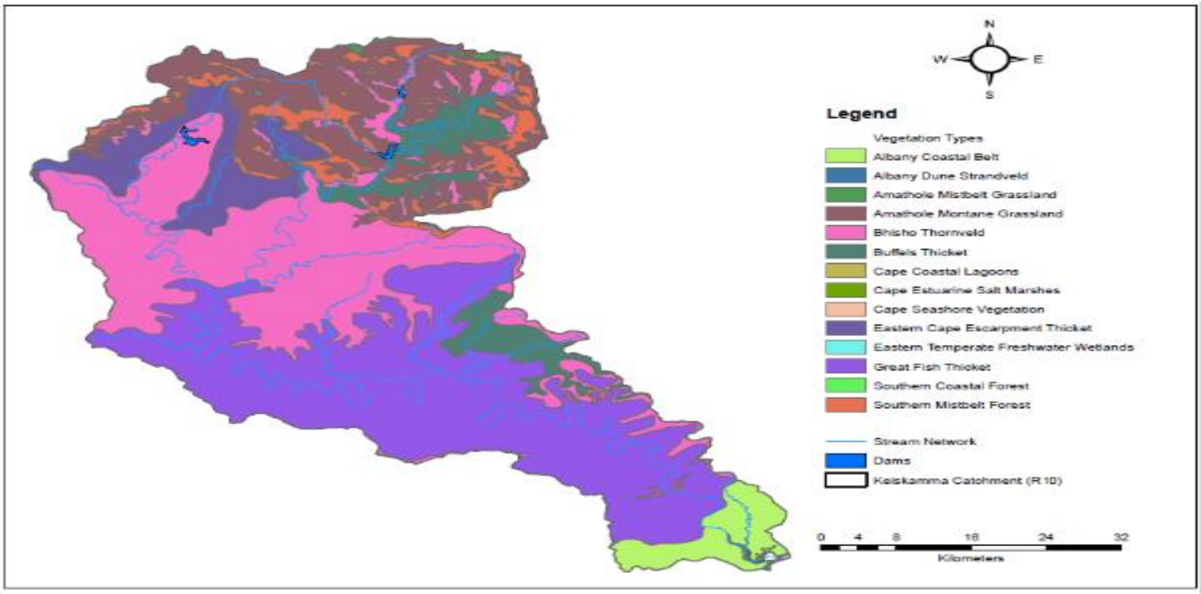

Figure 3.3. Vegetation types of the catchment

Source of vegetation classification: SANBI (2012)

Invasion by Invasive Alien Plants (IAPs) is a problem faced by the catchment. Invasion of Eucalyptus *spp.*, *Acacia Mearnsii* (Black Wattle) and *Acacia dealbata* (Silver Wattle) are noted throughout the catchment.

Heavy infestations of Wattle are particularly noted in the upper Keiskamma catchment sections. Stands of *Opuntia ficus-indica* (Prickly Pear) also occupy certain hillslopes of the catchment. The invaders are also found along the catchment's riparian vegetation but according to Mbikwana & Bushula (2008), riparian invasions are not yet serious problems for the catchment, except *Lantana and Sesbani sp.*

Central-coastal Keiskamma is dominated by degraded valley Bushveld (Mucina and Rutherford, 2006; Rutherford *et al.*, 2012). The vegetation type in these sections of the catchment forms part of the Bisho Thornveld, Buffels Thicket, Great Fish River Thicket and the Albany Coastal Belt vegetation units (SANBI, 2012). Mhangara *et al.* (2011) point out that these vegetation units, particularly in the central part of the catchment, are prone to fragmentation and degradation.

Abandoned land of the catchment is occupied by overgrazed and degraded poor grass species. Palmer and Avis (1994) observe that *Pteronia incana* (Blue bush) and *Elytropappus rhinocerotis* (Renosterbos) are shrub invaders occupying the catchment's disturbed and undisturbed hillslopes. Distribution of the species is associated with severe soil erosion (Kakembo and Rowntree, 2003; Kakembo, 2009; Manjoro *et al.*, 2012b). Similarly, the catchment's resident vegetation is prone to degradation by alien plant infestation, shrub encroachment, overstocking and overgrazing, wood gathering, unregulated fires and poor land and farming practices. Vegetation dynamics prevalent throughout the catchment play a key role in the hydrologic response of the catchment.

3.2.5 Soils

Story (1952) reveals that soil types of the area are reflective of the underlying geology. However, due to fluvial processes on the catchment's surfaces, certain soils have undergone displacement from their original environments. Owing to this displacement of soils and pedogenesis over time, the area appears to have diverse soil types as reported by previous studies conducted in the area (Malgas, 2008; Haindongo, 2009; Maphiri, 2009; Mhangara, 2011)

On the basis of Story (1952) Haindongo (2009) observe that the soils of the study areas are eutrophic greyish brown to shallow litholic soils. These soils are derived from the underlying geologies, sandstone, shale, and red mudstone which are part of the Beaufort and Ecca groups of the Karoo supergroup deposits of the Triassic and Permian periods (Haindongo, 2009; Maphiri, 2009). Other scholars point out that the area is also characteristic of deep red and yellow latosolic clays (Malgas, 2008; Maphiri, 2009).

Soils of the catchment tend to reflect patterns of physiography; well-drained soil tends to occur in old river terraces. Thin and shallow soils are prevalent on steep slopes whilst along drainage lines, soils with poor drainage occur (Haindongo, 2009). Several valleys of the catchment are characterised by nutrient rich mudstone derived soils. Other soils that occur in the catchment include greyish to brown and reddish soils. The heterogeneity of soil types occurring in the catchment is also essential in determining surface hydrological processes.

3.2.6 Geology

The Beaufort series of the Karoo supergroup are the major geological types that underlie the Keiskamma catchment (Mhangara, 2011). The Beaufort and Ecca series of the Karoo supergroup characteristically yield highly erodible sedimentary lithology ranging from shale, mudstone and sandstone (D’Huyvetter, 1985; Story, 1952; Mhangara, 2011). The catchment is predominantly underlain by sedimentary rocks including shale and mudstone geology. Granitic intrusions such as dolerite also occur within the catchment (see Figure 3.4). Owing to the problematic nature of the underlying geology, the topsoil is susceptible to erosion which is further exacerbated by vegetation degradation deterioration (Mhangara, 2011).

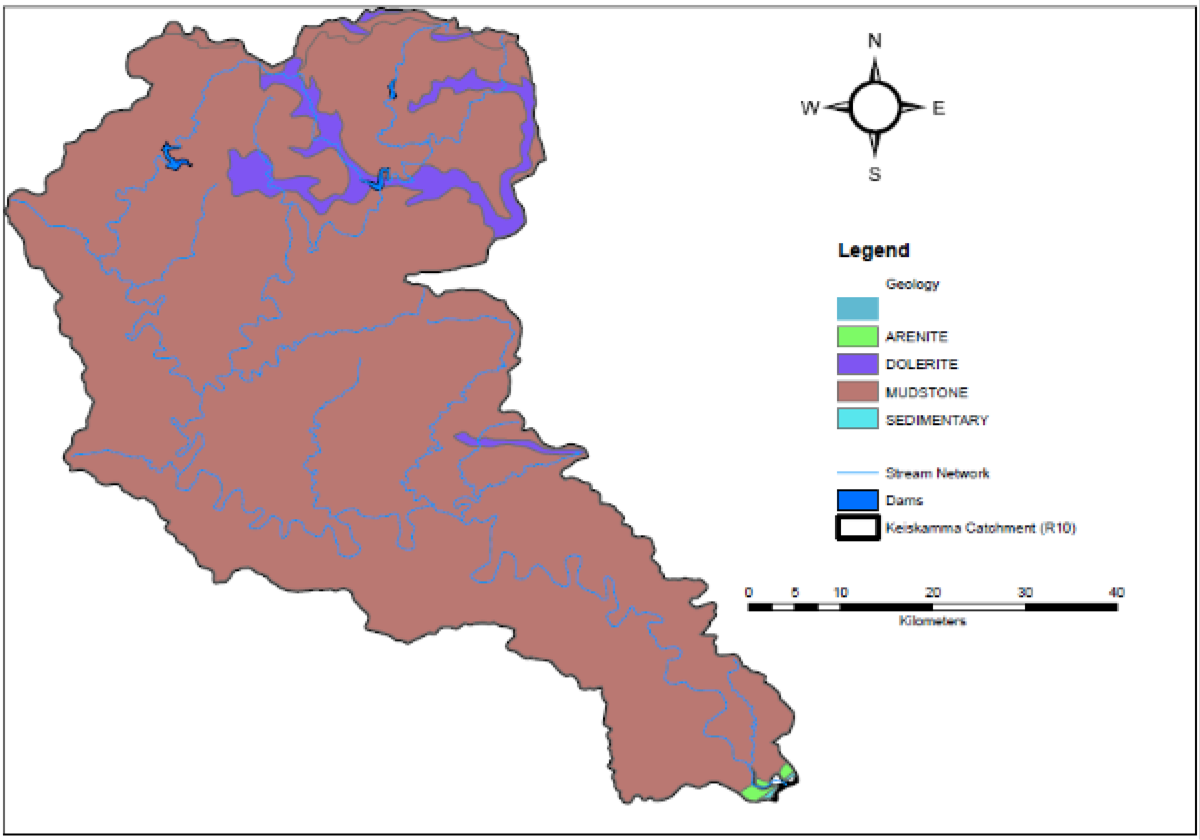

Figure 3.4. Geology of the catchment

3.2.7 Topographic characteristics

The Keiskamma catchment can be divided into three topographic (or eco-regions) zones; *viz* the escarpment zone on the headwaters, the coastal plateau knowns as the drought corridor occupying the middle reaches and coastal zone on the lower reaches (Mbikwana & Bushula, 2008). The upper watershed of the catchment, the Amatole mountains characterise the escarpment zone which is characterised by steep slopes and high elevations of up to 1,938 masl (see Figure 3.5). The escapement zone is also typified by distinctively high rainfall (Mhangara *et al.*, 2011). The Coastal Plateau lies between 600 and 900 masl, extends to the foothills of the Amatola mountain range and covers most of the Keiskamma Tertiary catchment (Mbikwana & Bushula, 2008). The coastal belt zone broadens from the coast into the watershed to about 20 km wide. The coastal plateau and the coastal zone are deeply incised and bisected by the Keiskamma river. The catchment is inherently a dendritic drainage pattern.

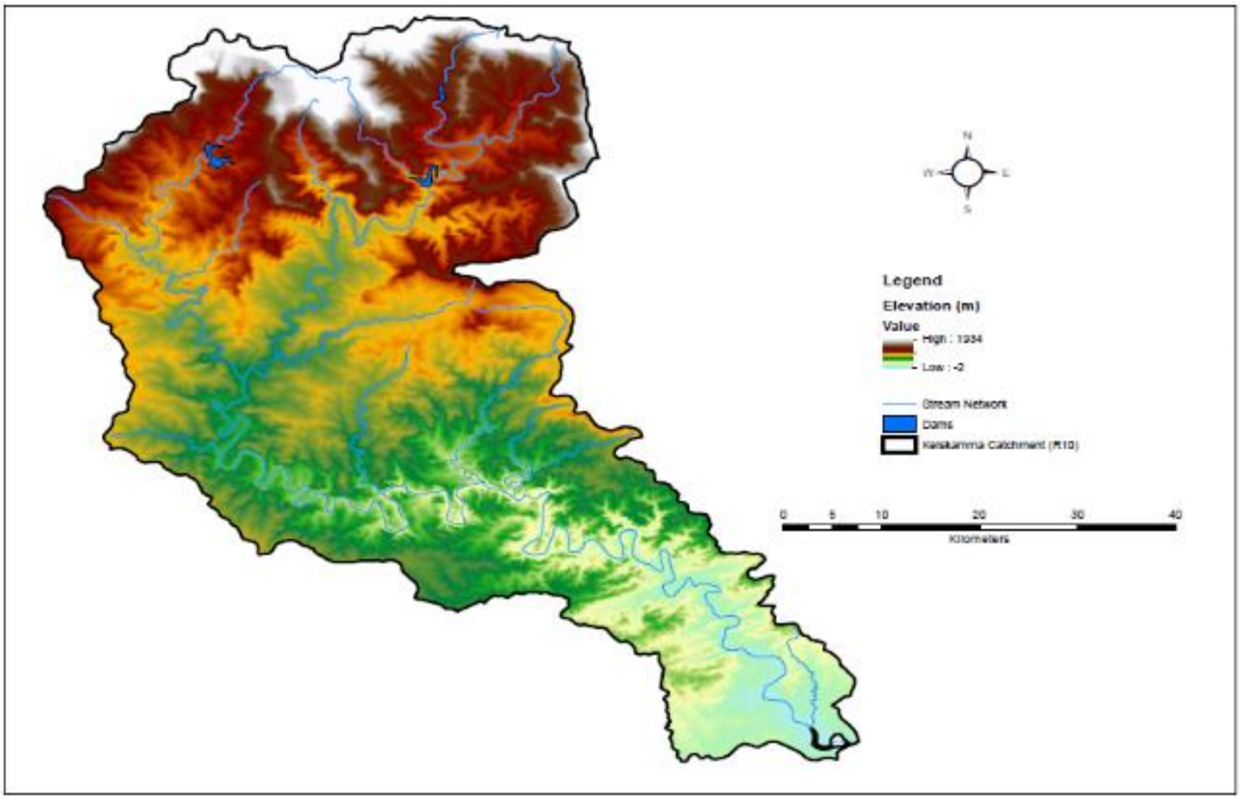

Figure 3.5. Elevation of the Keiskamma catchment

3.2.8 Demography and land use

The catchment accommodates several rural areas punctuated by a few small towns. The rural homesteads are characterised by isolated patterns. The catchment covers parts of three local municipalities; namely Amahlathi, Nkonkobe and Ngqushwa. The mean population of these municipalities has declined from 1996-2016 (StatsSA, 2016). In 1996 and 2016, the mean population of the municipalities was 388 743 and 328 094 respectively (StatsSA, 2016). This indicates a population decline of 60 649 over the 20 year period. From this trend, it can be deduced that the population is projected to register a decline, or very little growth, primarily because of lack of employment opportunities in the area coupled by rural-urban migration.

The major land use activities within the catchment include cultivated land, grazing land, abandoned formerly cultivated land, and residential areas/rural homesteads. These activities occupy the catchment's hillslopes adjacent the major rivers. The upper watershed of the catchment accommodates commercial forestry plantations, which occupy less than 100 ha on the Amatola mountains.

The major and active cultivated lands within the watershed are in Cata, KwaXhangwe, Lower Gxulu, NqoloNqolo, Ngqudela, Central Keiskammahoek, and Bomapass locations. According to Mbikwana & Bushula (2008), land cultivated under irrigation is less than 1500 ha. The largest irrigation schemes include the Keiskammahoek covering 854 ha and Ngqudela (356 ha). Based on satellite imagery observations, the University of Fort Hare dairy is also one of the largest irrigation schemes in the catchment.

As a result of the lack of judicious land management systems, the catchment's grasslands are prone to overgrazing and susceptible to soil erosion. Abandoned formerly cultivated lands are widespread throughout the catchment. These lands suffer serious degradation in the form of severe erosion and encroachment by IAPs, particularly *P. incana* (see Figure 3.6).

Figure 3.6. *P. incana* invaded and eroded hillslope

4 CHAPTER 4: METHODS AND MATERIALS

Introduction

This chapter presents the methods and techniques used for this investigation. Firstly, the methods and techniques used to assess land use/cover changes are presented. These detail the process of satellite imagery classification using a pixel-based supervised classification method. The section also details the accuracy assessment process of the classified images as a way to validate the degree of reliability of the produced land use maps for decision-making. The maps were then further used as land use maps inputs for the SWAT model.

Second, the process of modelling the catchment hydrological response using the SWAT hydrological model is explained. Calibration and validation are also explained in conjunction with model accuracy assessments. This section also explains the approach used to assess the effects of impoundments on the hydrological response of the catchment.

Third, field experiment methods used to determine hydrological impacts of *Pteronia incana* are outlined. Fourth, in response to one of the research questions of the study, methods used to assess the influence of rainfall on the hydrologic response are explained. The last section of this chapter explains statistical methods used in this investigation to analyse obtained data.

4.1 Assessment of land use/cover changes between 1994 and 2016

Multispectral imagery of 1994 and 2016 were obtained from the LANDSAT programme to identify and assess land use/cover (LULC) changes of the Keiskamma catchment. Assessment of LULC changes was important to track what has changed over time and to explain impacts of the changes on the hydrological response of the Keiskamma catchment. The LULC change mapping was also integral to produce land use/cover maps required by the Soil and Water Assessment (SWAT) hydrologic model, which was employed in this investigation to simulate the hydrological response of the catchment to LULC changes.

4.1.1 Image acquisition

Multispectral images from the Landsat programme were sourced from the United States Geological Survey (USGS) imagery archives. Landsat 5 Thematic Mapper (TM) images of 10 Feb 1994 and Landsat 8 Operational Land Imager (OLI) of 23 Feb 2016 were downloaded from the United States Geological Survey database. To minimise seasonal differences, images were deliberately chosen from the same season (Lu and Weng, 2007; Mhangara *et al.*, 2011). Figure 4.1 shows the colour composites of the images used in this study, where (A) presents the image of 10 February 1994 and (B) the image of 23 February 2016.

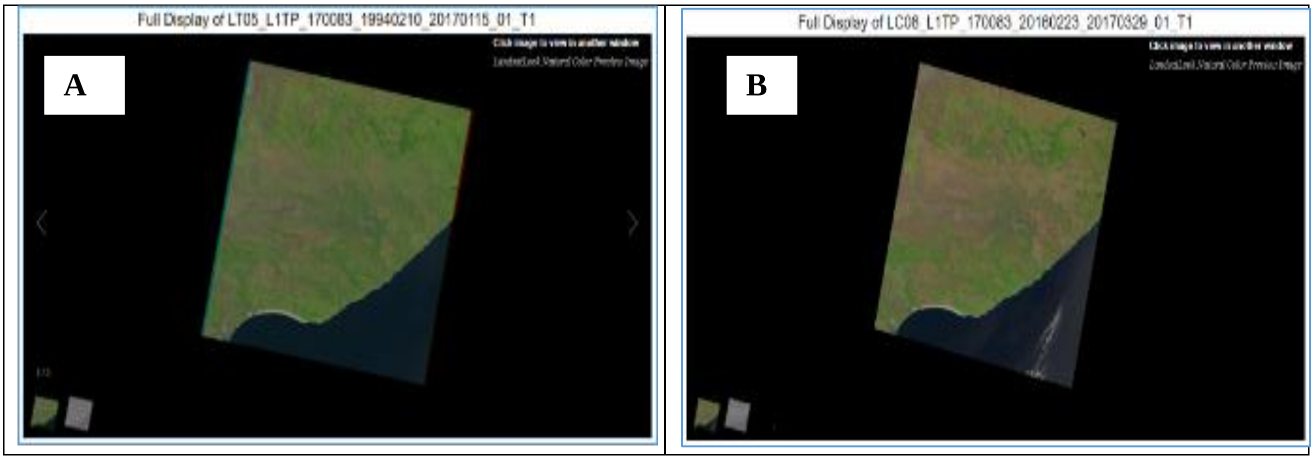

Figure 4.1. Colour composite of the study Landsat images

Table 4.1 shows details of the Landsat 5 Thematic Mapper (TM) image of 10 Feb 1994 and Landsat 8 OLI of 23 Feb 2016 used in this study.

Table 4.1. Details of classified images used in this investigation

Image ID	**Path**	**Row**	**Date of Acquisition**	**Landsat family**	**Resolution**
LT51700831994041JSA00	170	083	10 February 1994	Landsat 4-5 TM C1 Level-1	30 m
LC81700832016054LGN01	170	083	23 February 2016	Landsat 8 OLI/TIRS C1 Level-1	30 m

4.1.2 Pre-processing

According to Lu and Weng (2007), image pre-processing is an integral step towards a successful image classification process. Image pre-processing involves image restoration through correction of bad lines, image geo-registration, atmospheric correction, filtering, radiometric calibration and topographic correction. Conversion of image dataset formats to certain specific formats is also an important step of image pre-processing (Lu and Weng, 2007). The process of image pre-processing involves a combination of the aforementioned sub-steps depending on the research project's aim and specific objectives, and the quality of the imagery.

In the present study, the imagery was converted from GeoTIFF to IDRISI TerrSet format before further processing and analysis. Their initial GEOTIFF format is incompatible with IDRISI TerrSet hence the conversion to IDRISI TerrSet. The process of importing from GEOTIFF to IDRISI TerrSet was done for all the bands required for image classification. Bands 1, 2, 3 and 4, corresponding to the blue, green, red and near infrared bands for the Landsat 05 image and bands 2, 3, 4 and 5 for the Landsat 08 image were imported.

The images did not require any atmospheric correction because cloud free imagery was obtained. Radiometric correction, which is a requirement for datasets over multiple time periods, was not performed in this investigation since the study used two single-date images. Imagery data downloaded from USGS Explorer is by default geo-registered; hence, obviating the need to georeferenced imagery. To visualise the imagery bands information, image colour composites were created (see Figure 4.2).

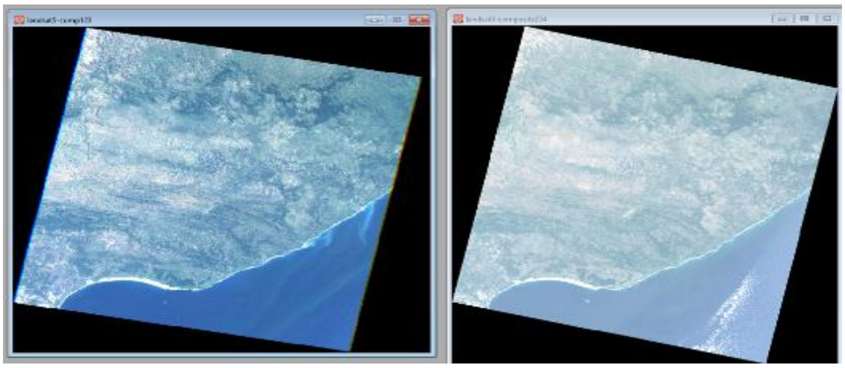

Figure 4.2. Image natural colour composites of 10 February and 23 February 1994 and 2016 respectively.

Sub-setting

After creation of the images' natural colour composites, subsets of the original images were created. The aim was to extract the spatial extent of the Keiskamma catchment. The study Area of Interest (AOI), which covers the catchment, was extracted from the imagery using Google Earth Pro observations simultaneously with TerrSet to derive a set of Ground Control Points (GCPs).

Image georegistration

As aforementioned, imagery data downloaded from USGS Explorer is by default geo-registered; hence, obviating the need to georeferenced imagery. Georeferenced imagery is imperative for the reliability of, among other things, image classification results. Classified images eventually undergo planimetric measurements for area calculations. To obtain accurate calculations, the classified images should have an appropriate projected coordinate system (PCS). In the present study, the Universal Transverse Mercator (UTM) was adopted as the PCS, and WG84 UTM_35S was the specific coordinate system employed.

4.1.3 Image classification

Classification of the Landsat images was done using the supervised image classification approach. The images were classified into six (6) classes according to the classification scheme developed by Anderson *et al.* (1976) for the interpretation of remotely sensed data. The classification classes included (1) Built-up areas, (2) Agricultural land, (3) Dense vegetation, (4) Rangeland (5) Bare and eroded land and (6) Water. Table 4.2 shows land use/cover types used in this investigation. The table also assigns the equivalent land use/cover change codes as required by the SWAT model which was employed in this study to model the hydrologic response of the catchment to land use/cover changes.

Table 4.2. Land use/cover types descriptions

ID Number	Land use / cover types	SWAT Classification Code	Land use /cover type description
1	Built-up areas	URLD	Includes both rural settlements and small towns within the catchment. However, the catchment is predominantly rural. Medium density residential areas are few. Hence, built-up areas of the catchment are categorized as Rural.
2	Agricultural land	AGRL	A combination of cropped and pasture land in the catchment. Includes both commercial and subsistence farms.
3	Dense vegetation	FRSE	Contains evergreen forests, dense riparian and thicket vegetation. These land cover types are noted by the dense green in satellite imagery.
4	Rangeland	RNGE	Categorises low lying vegetation, predominantly grassland and small shrubs thriving in the catchment.
5	Bare and eroded land	RNGB	Represents land that lacks vegetative cover, exposed and eroded.
6	Water	WATR	Groups together rivers and dams of the catchment

The first step in classifying the images was to select training data of the respective land use/cover change types. Selection of training sites is critical for successful image classification (Lu and Weng, 2007). According to Anderson *et al.* (1976), the basic rule is that training data should not be below ten times the number of bands to be used for the classification. In the present study a random stratified approach of selecting training sites per land use/cover change type was undertaken. Post training data categorisation, spectral signatures for the different land use/cover types, were created using the MAKESIG functionality in IDRISI TerrSet.

Image classification using the Maximum Likelihood (MAXLIKE) algorithm was adopted after signatures were developed for the respective land use/cover types. The Maximum Likelihood classifier was chosen for this investigation owing to its proven credibility in producing reliable image classification results (Lu and Weng, 2007; Reis, 2008; Conrad *et al.*, 2015). The classifier derives differences between the class means together with differences between the covariance matrices (Lu and Weng, 2007).

4.1.4 Classification accuracy assessment

Accuracy assessment evaluates the degree of reliability of classified maps. Accuracy assessment encompassed two major steps; *viz* sampling of Ground Control Points (GCPs) and accuracy assessment of the land use maps. Stratified-proportional random sampling of ground control points (GCPs) was the basis for validation of classification accuracy for both classified maps. GCPs were directly proportional to the size of the respective land use/cover classes.

Reference data in the form of GCPs for 1994 were taken from Google Earth imagery for 2016, using features visible on both dates. Google Earth imagery archives of the catchment for 1994 were of a very coarse spatial resolution so features on the ground were not visible. To sample GCPs, a point shapefile was created and assigned a projected coordinate system spatial reference (WGS84_UTM_35S) owing to the land use maps that were also in a projected coordinate system. GCPs were generated in ArcMap as random points for the respective land use/cover types and each land use/cover type was assigned a unique value that describes each respective land use/cover type (see Table 4.3).

Table 4.3. Unique values for GCPs of land use/cover types

Value	Land use/cover types
1	Built-up areas
2	Agricultural land
3	Dense vegetation
4	Rangeland
5	Bare and eroded land
6	Water

The final GCPs are shown in the figure below (see Figure 4.3)

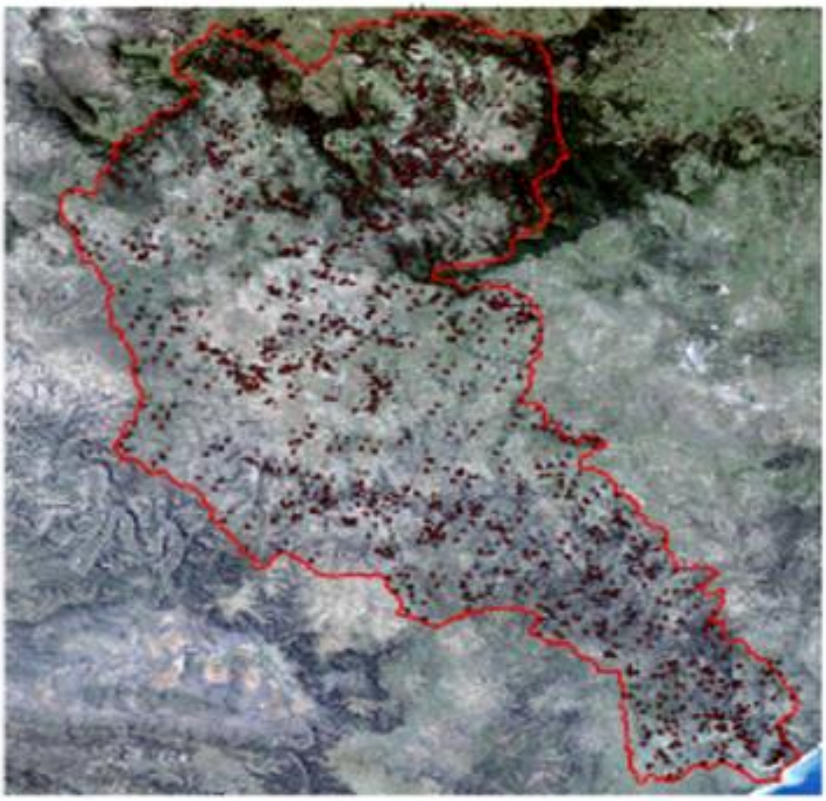

Figure 4.3. Ground Control Points (GCPs) of 1994 in ArcMap

GCPs used for accuracy assessment of the land use/cover map of 2016 were sourced from Google Earth imagery archives (see Figure 4.4). An image of February 2016 was used to select GCPs for an accuracy assessment of the classified land use/cover map of 2016.

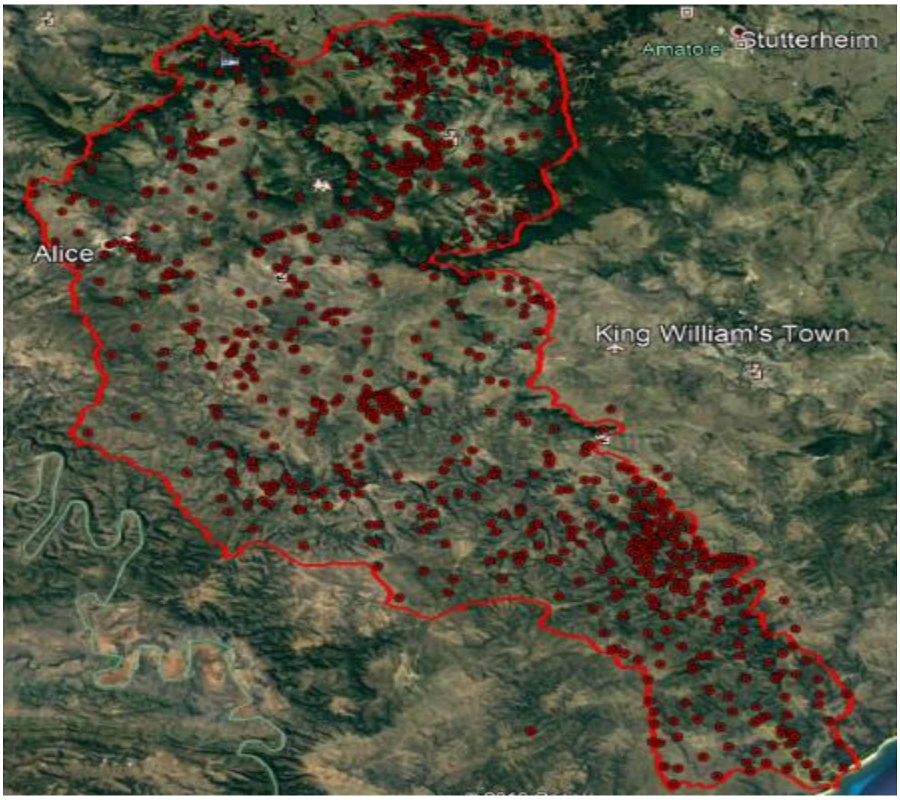

Figure 4.4. Ground Control Points (GCPs) of 2016 image in Google Earth Pro.

Ground Control Points (GCPs) that were sourced from Google Earth were in Keyhole Mark-up Language (KML) format, which is a file extension used to display spatial data in Google Earth. The GCPs in KML format were loaded and displayed in QGIS. Unlike ArcMap, QGIS displays KMLs directly without requiring any conversions. The GCPs were then converted into shapefile format to allow spatial analysis in ArcMap. The KML format GCPs had a default geographic spatial reference system as WGS84. During the conversion, the default spatial reference was redefined to a projected coordinate system, WGS84 UTM Zone 35S. The land use maps produced had WGS84 UTM 35S projected spatial reference. Therefore, to avoid incongruent overlays, it is important for all spatial layers to have the same spatial reference. The GCPs were then exported into ArcMap for the actual accuracy assessment process. The following table shows land use/cover types and the number of GCPs that were used to validate accuracy of the classification (see Table 4.5).

Table 4.5. Number of Ground Control Points per land use/cover types for years, 1994 and 2016

	Number of GCP	
Land use/cover types	**Image of 1994**	**Image of 2016**
Built-up areas	101	110
Agricultural land	193	145
Dense vegetation	315	283
Rangeland	317	291
Bare and eroded land	113	178
Water	95	94
TOTAL	**1134**	**1101**

Accuracy assessment of the land use/cover maps

To assess the accuracy of the land use maps, the GCPs were overlain with the classified maps that were exported from IDRISI Terrset to ArcMap 10.4. An error matrix was used to work out accuracy assessment of the classification. According to Anderson *et al.* (1979), an overall classification accuracy of 85% is acceptable. Images were iteratively classified until a satisfactory classification accuracy assessment percent was achieved. The Kappa coefficient, which is a multivariate statistic, was also used to further assess the accuracy of the agreement between values of the classified map and the ground truth/ reference data. Users' and Producers' accuracy as well overall classification accuracy was determined.

Extraction of land use/cover types values to points

After overlaying GCPs with a land use/cover map, the spatial analysis tool called Extract Values to Points was used. The tool assigns raster values of specific land use/cover types to GCPs with the land use/cover type value. The operation produces a point feature (shapefile) with an attribute table that shows agreement of the GCPs with the raster values of the specific land use/cover types.

Frequency analysis of land use/cover types values to GCPs

Frequency analysis of the GCPs to raster values of the land use/cover types was done. The frequency table, under the filed FREQUENCY, indicates the number of GCPs that were assigned to the correct raster values of each LULC class. The frequency table also indicated the number of GCPs that were assigned to the wrong LULC types (see Figure 4.6). Basically, the frequency table shows the number of GCPs that agree and do not agree with the classified land use map.

Table

Extract_Points_Frequency1

OBJECTID *	FREQUENCY	Lulc	RASTERVALU
1	89	1	1
2	20	1	4
3	1	1	5
4	1	2	<Null>
5	99	2	2
6	12	2	3
7	18	2	4
8	6	2	5
9	3	3	2
10	126	3	3

Figure 4.6. Frequency table accuracy assessment

Generation of an Error Matrix

An error matrix was computed using the Pivot Table. The pivot table analysis tool computationally arranges the frequency table into an error matrix format (Figure 4.7) using the fields; *viz* Frequency, GCPs and Land use raster values. The error matrix was exported to Microsoft Excel for further analysis.

Table

Extract_Points_Frequency1_Pi4

OBJECTID *	Lulc	RASTERVALU1	RASTERVALU2	RASTERVALU3	RASTERVALU4	RASTERVALU5	RASTERVALU6
1	1	89	0	0	20	1	0
2	2	0	99	12	18	6	0
3	3	0	3	126	0	0	0
4	4	0	3	10	217	1	0
5	5	0	0	0	27	151	0
6	6	0	0	2	10	0	82

Figure 4.7. Error matrix generation using Pivot table analysis tool

4.1.5 Change detection and quantification

Quantification of the area of the respective land use/cover types was done in ArcMap. After images were classified in IDRISI Terrset, they were saved as raster files and opened in ArcMap. A shapefile of the Keiskamma tertiary catchment (R10) was used to extract the AOI from the classified images. This was important because the classified maps included areas outside the actual study area. Extracting the AOI ensured that change quantification only accounted for land use/cover types exclusively within the catchment of study.

The areas of the respective land use/cover types were then calculated. The areas of LULC types were calculated by multiplying the number of pixels for each land use/cover type by the imagery pixel spatial resolution of 900 m^2 (i.e. 30 m x 30 m) and the areas were converted into hectares by diving by 10 000, since 1 hectare is equivalent to 10 000 m^2 (Odindi *et al.*, 2012b). LULC was calculated using the following formula:

$$LULC\ Change = Present\ LULC\ type\ (ha) - Past\ LULC\ type\ (ha)$$

Where: Present LULC type (ha) refers to the size of the respective LULC type for the year of 2016 and Past LULC type (ha) represented the size of each LULC type for 1994.

As aforementioned, the LULC change mapping was imperative to produce land use/cover maps required by the Soil and Water Assessment (SWAT) hydrologic model. The SWAT model was used in this study to simulate the hydrological response of the Keiskamma catchment to land use/cover changes. Amongst the model inputs, catchment land use maps are mandatory during both model calibration and validation. In the present study, land use/cover map of 1994 and 2016 were used for model calibration and validation respectively.

4.2 Modelling hydrological responses of the catchment using SWAT

Land use/cover changes in the Keiskamma catchment have been investigated by several studies (Rowntree and Dollar,1994; Haindongo, 2009; Mhangara *et al.*, 2011). However, empirical evidence of hydrological impacts at catchments scale remains not fully understood. Therefore, it is imperative to assess the hydrologic response of the catchment to land use/cover changes, using a distributed basin-scale hydrologic model. Distributed basin-scale hydrologic models simulate hydrologic processes for the whole catchment accounting for spatial heterogeneity.

The SWAT hydrologic model employed in the present study is a physically based, semi distributed and process based hydrologic model which was developed and designed to predict land use and management impacts on water at catchment scale (Arnold *et al.*, 2012). The land use maps produced from image classification were used by the SWAT model as land use inputs to simulate hydrological responses for the specific periods. Below is a schematic illustration of the hydrological modelling process followed in this study (see Figure 4.8)

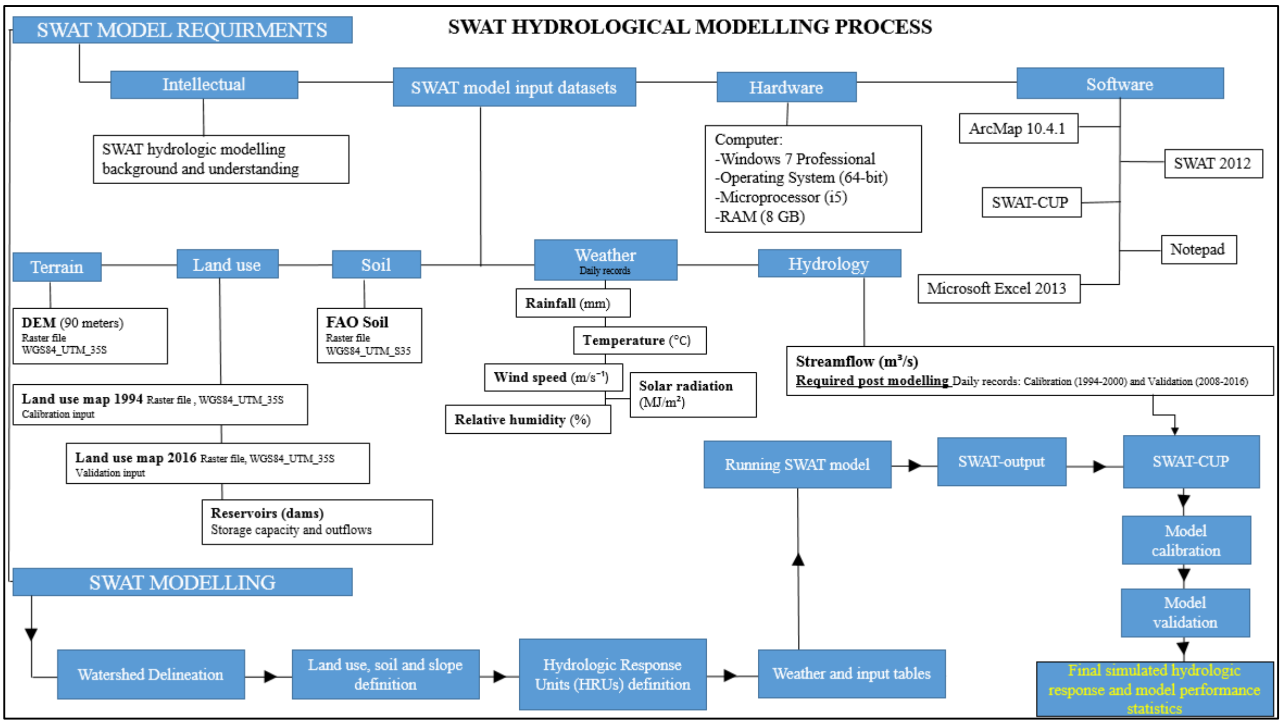

Figure 4.8. SWAT hydrological modelling process schematic illustration

4.2.1 SWAT input datasets

The basic data requirements of SWAT include topographic, meteorological and land cover/ use data. There are many other input datasets for SWAT but they are optional (Neitsch *et al.*, 2002; Gassman *et al.*, 2007; Arnold *et al.*, 2012a). Stream flow data is a required input after the modelling process to compare the simulated streamflow with the observed data during model calibration and validation.

Topographical data

A Digital Elevation Model (DEM) of 90 m (3 arc-seconds) for the catchment was obtained from The Shuttle Radar Topography Mission (SRTM). A shapefile of the catchment was used to mask the area of interest from the DEM in ArcMap. The DEM defines the topography of the catchment by describing the elevation of the catchment at any point at a specific spatial resolution. The DEM was used for watershed delineation, analysis of the watershed drainage pattern, streams and slope.

Meteorological data

SWAT model weather data requirements include daily precipitation, maximum and minimum air temperature, solar radiation, wind speed and relative humidity. The weather data required by the SWAT model can be records of observed data or the model can simulate them. In cases where daily records were missing, the value "-99" was used as per the SWAT weather data documentation guide (Srinivasan *et al.*, 2005)

In the present study, daily measurements of precipitation, maximum and minimum air temperature, wind speed, relative humidity, were obtained from the South African Weather Services (SAWS). Solar Radiation is not measured by weather stations near the Keiskamma catchment and therefore, solar radiation data were simulated by the Global Weather Generator.

No weather stations are located exactly in the catchment. Two weather stations and two rainfall gauges were used as sources of weather data. The weather stations used in this study are located at KwaNoncampa (Bisho) and Fort Beaufort, 5 and 20 km east and west of the central catchment boundary respectively.

The assumption made in the present study is that weather and indeed climate conditions captured by the two stations have a strong influence on the hydrological conditions of the central Keiskamma catchment and surrounding areas.

The two rainfall gauges located in the upper part of the catchment are Hogsback and Amatole. For the rainfall stations, weather data elements such as temperature, wind speed, relative humidity and solar radiation were sourced from the Global Weather Generator. The rainfall conditions captured by the two stations are an important input into the catchment's hydrology. Weather conditions captured by the weather stations and rainfall gauges are assumed to be representative of the entire catchment.

Land use/cover data

The study used two land use/cover maps that were produced by image classification, as detailed in the image classification section. The maps were for the years 1994 and 2016. The former land use/cover map was an input during the calibration period and the latter was used during model validation. A lookup table was created to link the land use map classes to the SWAT model recognised land use/cover codes.

Soils data

The Harmonised World Soil Database (HWSD.mdb) and soil map was downloaded from the Food and Agriculture Organisation (FAO) (2012). The HWSD encompasses world soil information which can be identified by various mapping and soil type codes. The world soil map was in raster format. A shapefile boundary of the Keiskamma catchment was used to extract the soil data of the catchment from the world soil map in ArcGIS with values representing soil types known as Global Mapping Units. The output attribute table showed that the catchment has 20 different soil types.

For the SWAT model to recognise the catchment's soils, the data extracted from the HWSD map was linked to the ArcSWAT user-soil database (**SWAT2012.mdb**). This was achieved by downloading a custom SWAT2012.mdb provided online by Gungor (2014). The database provided by Gungor contained soils of the world with all their physical and chemical properties in a format required by the SWAT model.

The HWSD.mdb has the columns named "MU_GLOBAL" and SU_CODE90, which are the Global Mapping Unit and the numerical code for the FAO-90 soil classification system respectively (Nachtergaele, 2009). The two are important identifiers that provide the link between the GIS layer and the attribute database for world soils. SWAT2012.mdb does not contain MU_Global and SU_CODE90. Nevertheless, when the two mapping units are read in HWD.mdb under MU_GLOBAL, the SU_CODE90 has a value for specific soil types which correspond to the column "SEQN" in SWAT2012.mdb. The SEQN soil identifiers were linked to SWAT soil names (SNAM). Therefore, soil name identifiers under "SNAM" were used to create a user-lookup table for the catchment soils.

Each unique soil value was searched under MU_Global to identify its SU_CODE90 which ultimately links the soil to SEQN on the SWAT soils database. A user lookup table integrating the catchment soil's raster and SWAT was created; where the value represented the soil identifier in the raster and name represented soil type as found in the SWAT database. The soils of the Keiskamma catchment are mainly Clay Loam, Loam, Clay and Sandy Loam and the majority of the soils fall within hydrologic group D with a fraction under group C (see Table 4.4)

Table 4.4. Keiskamma catchment soils information

Raster Value	SWAT Code	Soil Name	Hydrologic Group
29646	Jc23-2a-103	Clay Loam	D
29837	I-R-74	Loam	D
29848	Ne12-3b-156	Clay	D
29864	Jc32-2a-115	Loam	D
29886	Je31-2-3a-123	Clay Loam	C
29888	Lg35-1a-144	Sandy Loam	C
29912	I-62	Loam	C
29928	Jc26-2-3a-106	Clay Loam	D

Hydrological data

Streamflow is an input required for calibration and validation of simulated streamflow. Streamflow data of the catchment for the period 1994-2016 were obtained from Department of Water and Sanitation (DWS). The catchment has a flow gauging station downstream near Hamburg located at 33°11'6.63"S, 27°23'26.30"E (see Figure 4.9). The flow gauging station has been active since 01 August 1969.

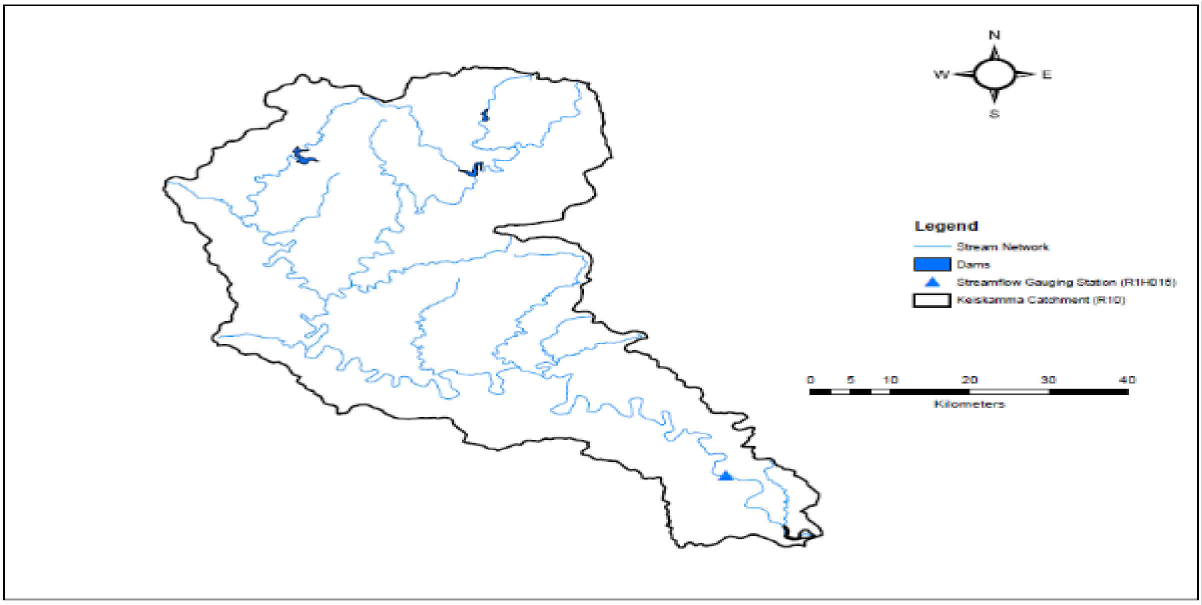

Figure 4.9. Streamflow gauging station location

Streamflow datasets from 1994-2000 and 2010-2016 were used for SWAT model calibration and validation respectively (see Figure 4.10). Selection of streamflow data was informed by the availability of weather data for the same periods as well as avoidance of data gaps.

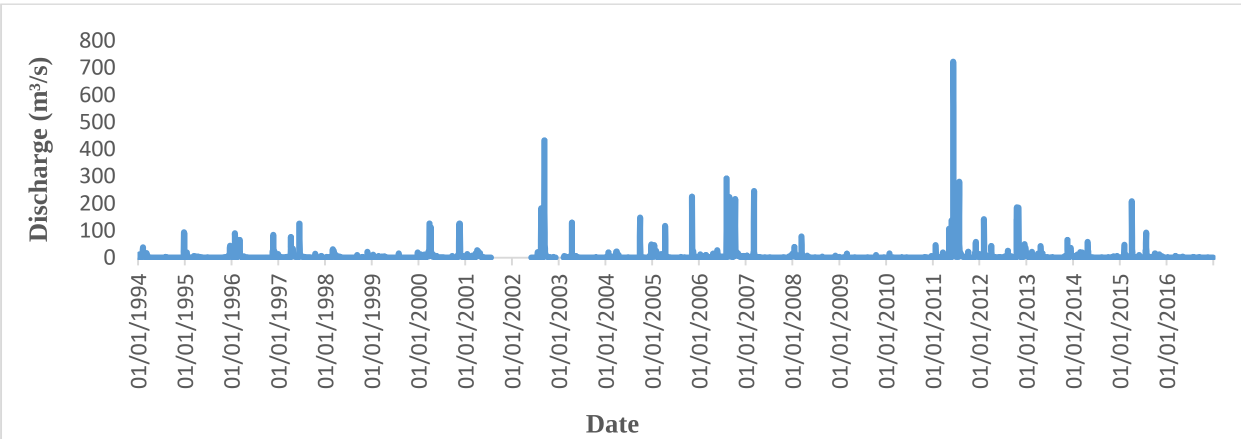

Figure 4.10. Streamflow records of the Keiskamma catchment from 1994-2016

4.2.2 SWAT model setup and modelling

Project set-up

Project set up was the first and necessary step for using ArcSWAT 2012 which ensured that SWAT extensions in ArcMap 10.4 are activated. This activated the ArcSWAT tool and defined a directory for saving of all project databases. Spatial Analyst ArcMap extension was also activated. SWAT Project Manager and SWAT Watershed delineator are the other two extensions that were initially enabled in ArcMap for the functioning of the ArcSWAT tool (Neitsch *et al.*, 2002).

Watershed delineation

Watershed delineation is an important step that uses a DEM to define the catchment boundary, size sub-basins, flow direction and accumulation, stream network and outlet points. Streams were defined using the automated flow direction and accumulation computation. The computation uses DEM information to define flow direction and accumulation in the catchment.

Size of the sub-basins of the catchment was delineated according to a suggested threshold area range in SWAT. The threshold area for each sub-basin of the catchment defines the degree of spatial delineation of the catchment, stream network detail, size and number of sub-basins within a catchment (Srinivasan *et al.*, 2005). Furthermore, the threshold sub-basin area defines the drainage areas necessary for the formation of stream origin or beginning (Neitsch *et al.*, 2002). In the present study the sub-basin threshold area was defined as 1221 ha for each sub-basin. The satisfactory threshold area was obtained through iteratively adjusting the threshold and stream network definition until a satisfactory stream network was achieved.

Outlets of the sub-basins were automatically located by the model. However, the catchment's main outlet was added to the location of the streamflow gauging station obtained from the DWS. Impoundments located in the catchment; namely Cata, Mnyameni, Binfield, Sandile, Debe and Dimbaza dams were also added manually during catchment delineation.
Reservoir information pertinent to volume, capacity and outflows as per SWAT documentation was obtained from DWS.

However, information about Mnyameni and Dimbaza was not available as they were not registered under DWS. Based on the delineated watershed, calculation of watershed parameters was done. According to Neitsch *et al.* (2002), this step calculates parameters for all sub-basins of the catchment. The function also assigns the mandatory sub-basin identification codes.

Land use, soil and slope definition, and overlay

After watershed delineation, land use, soil and slope information of the catchment was loaded into the model. Land use, soil and slope of the catchment are critical factors that regulate catchment hydrological responses. They were defined under the Hydrologic Response (HRU) analysis. According to Gassman *et al.* (2007), HRUs are the smallest spatial units of the catchment characterised by homogeneous topography, soil properties, slope and land use.

Land use

The land use map of 1994 was loaded from the project folder into the model. The grid field which defines the unique land use/cover classes was the VALUE field of the attribute table of the map. A lookup table was loaded to link the land use map classes with SWAT model land use classification. Rangeland (RNGE) was subdivided into grassland and shrubland (RNGB). The study's classification scheme defines rangeland (RNGE) as a mosaic of grassland and shrubs. However, according to the SWAT model, RNGE refers to only to the grassland component. To mimic rangeland conditions of the Keiskamma catchment, it was important to subdivide RNGE into grasses and shrubs as these two coexist within the catchment. Furthermore, splitting of the land cover RNGE permitted the capture of different hydrological processes at play within grasses and shrubs. After the refinement of the rangeland land use component, the land use map was then reclassified to integrate into SWAT land use codes. The 1994 map was an input map for the model calibration period of 1994-2000. The 2016 map was used as an input land use for the validation period of 2010-2016 following the same procedure.

Soils

A soil map of the catchment was loaded into the model. The soil's field "VALUE" was used as the field that defines the catchment's soil type as per the soil grid attribute table. A lookup table was used to link the soil grid into SWAT user soils database. The soils were then reclassified to effect integration of the grid soils into SWAT soil codes.

Slope

Slope is an important determining factor of the movement of water, sediment and nutrients within the catchment (Srinivasan, 2012). The Keiskamma catchment has a very variable topography. Therefore, slope discretisation was done using the Multiple Slope option. Slope of the catchment was categorised into five classes; namely 0-10%, 10-20%, 20-30%, 30-40% and >40%. The five classes account for slope diversity of the catchment. The created slope classification was then reclassified. Land use, soil and slope information computed into the model was then overlaid.

Hydrologic Response Unit (HRU) definition

HRUs are defined as areas of homogeneous land use, soils and slope. HRUs are integral catchment portions that determine the hydrological responses of the specific sections within the catchment which all sum up into overall catchment hydrological responses. The multiple HRUs option was employed in this investigation to define the Keiskamma catchment HRUs. The option sub-divides the catchment into areas comprising unique combinations of land use, soil and slope. The HRUs are the basis for prediction of processes such as surface runoff which are further routed to total surface runoff of the catchment (Srinivasan, 2012). The multiple HRUs option increases model accuracy of the water balance physical description (Srinivasan, 2005).

HRU thresholds define the minimum percentage of land use, soil and slope considered negligible to define formation of HRUs within sub-basins. The default HRU Definition option was used to define HRU thresholds. For this investigation, all thresholds were set to zero (0 %) for land use, soil and slope percentage over sub-basin area. The HRUs were then created on this basis. This approach is very important because the model simulates hydrological responses for every unique combination of land use, soil and slope within the catchment. Further, the land use refinement option was used because during the classification, grasses (RNGE) and shrubs (RNGB) were all classified as rangelands (RNGE). However, on the ground, the two land covers have unique hydrological responses. Cognisance of that resulted into splitting the land use RNGE into grassland (RNGE) and shrubs (RNGB).

Weather data definition and writing input files

Weather data is one of the critical inputs for the SWAT model. SWAT requires daily data records for: rainfall, minimum and maximum temperature, wind speed, solar radiation and relative humidity. These weather elements play a fundamental role in driving hydrological responses.

To feed weather data into the model, the location of the Weather Generator Data (source of weather data) was specified as a WGEN_user, which is a database that contains long-term weather statistical parameters and information about the catchment's weather stations. Location files for rainfall, temperature and wind speed data stations were loaded under the respective tabs. The station's location files contained information on the identification number, station name, latitude, longitude and elevation of the weather or rainfall station per weather elements.

Solar radiation and relative humidity data were simulated by the model. The weather stations used in this investigation do not measure solar radiation and owing to data gaps in relative humidity data, the simulation option was chosen. The SWAT model simulated the weather parameters using the embedded global weather generator function which simulates the missing data using the nearest weather stations to the catchment. After defining weather data, all SWAT input tables were generated. The model used an automated process to build input tables specific to the catchment into the SWAT2012.mdb.

Running the SWAT model

After successfully writing SWAT input files, the model was run to simulate hydrological responses of the catchment for the specific time periods. For the calibration period, SWAT was run from 01 January 1994 to 31 December 2000. A warm up period of 2 years was used as suggested by the SWAT user manual (Arnold *et al*., 2012a). The warm-up period is necessary for optimising model predictions. The model was then validated for the period 01 January 2010 to 31 December 2016. Different land use/cover maps were used for both calibration and validation to ensure that weather/climatic conditions correspond to the actual land use/cover conditions of the catchment for the specific periods. This permitted drawing inferences about hydrological impacts of land use/cover conditions that are physically accounted for and included in the model.

Model output (Hydrologic responses)

The modelled hydrologic response was imported and viewed from the SWAT output database. Several simulated hydrological processes are contained in the SWAT output database for HRUs, sub-basins and the catchments' main outlet. In the present study, streamflow (m^3/s), surface runoff (mm), soil moisture content (mm), evapotranspiration (mm), groundwater (mm) and sediment loss (tonnes) were the hydrological responses of interest. Simulated streamflow was encompassed by the main channel output database (output.rch) whereas surface runoff, soil moisture, evapotranspiration, groundwater and sediment loss were contained in both the HRU (output.hru) and sub-basin (outut.sub) output databases.

To assess the reliability of the simulated hydrological response, the predicted streamflow was compared with the observed streamflow. Agreement between the predicted streamflow versus the observed determined the reliability of the simulated hydrological responses. In assessing this agreement as an indicator of reliability of the simulations, calibration and validation were undertaken.

Assessing the effect of impoundments on streamflow

To assess the impacts of impoundments on the catchment's streamflow, discharge simulated in a sub-basin upstream was compared with discharge simulated for downstream sub-basin impoundments. This was validated using streamflow measured in sub-basins upstream and downstream of the dams. The comparison of upstream and downstream streamflow was done using hydrographs.

4.2.3 SWAT calibration, validation, performance assessment and uncertainty analysis

Introduction

A comprehensive set of statistics was used to assess the SWAT model performance and uncertainty analysis for both calibration and validation stages. The investigation used hydrographs, standard regression, dimensionless and error index statistics to assess SWAT model performance. Uncertainty analysis was undertaken using two statistics; namely p-factor and r-factor.

Hydrographs were the graphical method employed in this investigation for visual model performance assessment. Model bias, peak flows timing and magnitude differences and recession curves shape can be identified on a hydrograph (NRCS, 1986; USDA, 1986; Moriasi *et al.*, 2007)

Standard regression statistical methods are important for determining the degree of linearity of the relationship between simulated and observed data. Two standard regression statistics were employed in this study, the coefficient of correlation (R) and the coefficient of determination (R^2). The Pearson correlation coefficient quantifies the linear relationship between observed and simulated data whereas the coefficient of determination reflects the proportion of the variance in measured data (Moriasi *et al.*, 2007). The investigation employed one dimensionless statistic, the Nash Sutcliffe Efficiency (NSE).

Error indices measure the deviations in the units of the data of interest (Legates and McCabe, 1999; Moriasi *et al.*, 2007). Percent Bias (PBIAS) and Root mean square error (RSR) were used. Lower RSR values are commonly indicative of good model performance.

Percentage Bias (Pbias) which quantifies the mean tendency of the simulated data to be larger or smaller than the measured data, was used in this study. PBIAS values of low magnitude are indicative of accurate model predictions (Moriasi *et al.*, 2007). The mean and standard deviation of the simulated flow against the observed flow were other statistics used to assess the performance of the SWAT model. It is important to add the importance of simulated and observed streamflow.

The model performance evaluation used in this investigation was applied in line with the recommended statistics for general model performance ratings for a monthly time step simulation, emulating Moriasi *et al.* (2007) (see Table 4.5).

Table 4.5. Recommended model evaluation statistics for monthly time step simulations

Performance Rating	RSR	NSE	PBIAS (%)		
			Streamflow	Sediment	N, P
Very good	$0.00 \leq RSR \leq 0.50$	$0.75 \leq NSE \leq 1.00$	$PBIAS \leq \pm 10$	$PBIAS \leq \pm 15$	$PBIAS \leq \pm 25$
Good	$0.50 \leq RSR \leq 0.60$	$0.65 \leq NSE \leq 0.75$	$\pm 10 \leq PBIAS \leq \pm 15$	$\pm 15 \leq PBIAS \leq \pm 30$	$\pm 25 \leq PBIAS \leq \pm 40$
Satisfactory	$0.60 \leq RSR \leq 0.70$	$0.50 \leq NSE \leq 0.65$	$\pm 10 \leq PBIAS \leq \pm 25$	$\pm 30 \leq PBIAS \leq \pm 55$	$\pm 40 \leq PBIAS \leq \pm 70$
Unsatisfactory	$RSR > 0.70$	$NSE \leq 0.50$	$PBIAS \leq \pm 25$	$PBIAS \leq \pm 55$	$PBIAS \leq \pm 70$

Model calibration

The model was calibrated using SWAT-CUP which is a method that uses automatic calibration and uncertainty analysis. Sequential Fitting Uncertainty (SUFI-2) was the algorithm used to calibrate the model. SWAT-cup also includes parameter sensitivity and uncertainty analysis. The calibration procedure followed was per the SWAT-CUP user manual. The model was calibrated using observed streamflow records of the Keiskamma catchment outlet from 1994-2000 (Moriasi *et al.*, 2007; Arnold *et al.*, 2012b). For calibration, the target number of iterations for successful calibration is five (5) iterations each consisting of 250-500 simulations (Neitsch *et al.*, 2002; Moriasi *et al.*, 2007; Arnold *et al.*, 2012b). In the present study, each iteration consisted of 500 simulations and sixteen (16) catchment parameters were optimised (see Table 4.6).

The calibration parameters and parameter value ranges were advised by SWAT model experts from the SWAT-CUP online Google Groups. Model performance was assessed using the aforementioned comprehensive statistics.

Table 4.6. SWAT model calibration parameters' information.

Parameter name	Parameter abbreviation	Optimization method	Parameter value	
			Minimum	**Maximum**
SCS runoff curve number	CN2.mgt	Relative (r)	-0.2	0.2
Baseflow alpha factor (days)	ALPHA_BF.gw	Replace (v)	0	1
Groundwater delay (days)	GW_DELAY.gw	Replace (v)	0	450
Threshold depth of water in shallow aquifer required for return flow to occur (mm)	GWQMN.gw	Replace (v)	0	1500
Deep aquifer percolation fraction	RCHRG_DP.gw	Replace (v)	0	1
Threshold depth of water in the shallow aquifer for "revap" to occur (mm)	REVAPMN.gw	Replace (v)	0	1000
Groundwater "revamp" coefficient	GW_REVAP.gw	Replace (v)	0	1
Available water capacity of the soil	SOL_AWC.sol	Relative (r)	-0.25	0.25
Moist bulk density	SOL_BD.sol	Relative (r)	-0.25	0.25
Depth from soil surface to bottom of layer	SOL_Z.sol	Relative (r)	-0.25	0.25
Manning's "n" value for the main channel	CH_N2.rte	Replace (v)	0	0.4
Effective hydraulic conductivity in main channel alluvium	CH_K2.rte	Replace (v)	0	200
Soil evaporation compensation factor	ESCO.hru	Replace (v)	0	1
Plant uptake compensation factor	EPCO.hru	Replace (v)	0	1

Parameter name	Parameter abbreviation	Optimization method	Parameter value	
			Minimum	Maximum
Maximum canopy storage	CANMX.hru	Replace (v)	0	8
Surface runoff lag time	SURLAG.bsn	Relative (r)	-5	5

Nash Sutcliffe Efficiency (NSE) was the main objective function as it is commonly used as the main model performance indicator (Moriasi *et al.*, 2007; Adeogun *et al.*, 2014; Dos Santos *et al.*, 2014; Gyamfi *et al.*, 2016; Sun *et al.*, 2017b). Other statistics included Coefficient of Determination (R^2), Percent Bias (PBIAS) and Root mean square error (RSR). On the 4th iteration, "Good" model performance was achieved, and calibration was ended as the 5th iteration model performance was not good. Thus, the model was calibrated until "Good" model performance indicators were achieved.

Model uncertainty analysis

The p-factor and r-factor statistics embedded in SWAT-Cup were also used to assess SWAT model performance as well as uncertainty analysis for both calibration and validation. SWAT-Cup quantifies uncertainty using the 95% Prediction Uncertainty (95PPU) band. The p-factor is defined as a measure of the percentage of observed data falling within the 95PPU band. The p-factor index is a measure of SWAT model's capability to capture uncertainties. According to Arnold *et al.* (2012b), the p-factor should be 1, meaning that 100% of the observed data is within the 95PPU band. This implies that the model has taken into account all the correct processes. On the other hand, the r-factor is the thickness of the 95PPU band and the calibration quality. "The ideal value for the r-factor should be near 0 indicating high model performance. The combination of the p-factor and r-factor together indicate the strength of the model calibration and uncertainty assessment, as these are intimately linked" (Arnold *et al.*, 2012b, page 1496). In the present study, the p-factor and r-factor were used with the aforementioned model performance evaluation statistics to assess model performance and model uncertainty.

Validation

After achieving a satisfactory model performance during calibration, the model was validated for the 2010-2016 period using observed streamflow for the respective period. During the validation period, one iteration of 500 simulations was performed using model optimised parameters obtained on the 4th iteration of calibration to validate the model for the 2010-2016 period. After the iteration, model parameters were not further adjusted but used to report on model performance and uncertainty for the validation period.

4.3 Determining hydrological impacts of *Pteronia incana* (*P. incana*) invasion

An experimental set up was installed to investigate hydrological implications of *P. incana* invasion, surface runoff, and sediment loss and soil moisture retention dynamics. This was important in order to gain insights into the hydrological implications and impacts of the invader for hillslopes and ultimately catchment hydrology, as well as to support the SWAT hydrologic model.

4.3.1 Experimental set-up

An experimental set-up was conducted at hillslope scale to investigate hydrological impacts of the invader shrub, *P. incana.* Variables monitored at hillslope scale were soil moisture, surface runoff and sediment loss and were monitored from the 22 September 2018 to 09 November 2018 as the rainfall season was targeted. The invader shrub hydrological impacts were compared to grassland and eroded surfaces. Surface subunits prevalent in *P. incana* encroached landscapes is the invader, bare inter-patches and grassland. Noteworthy is that *P. incana* invasion is characterised by two surface conditions with unique hydrological and ecological states. The plant's tussock, also called the resource island, differs from the inter-patch areas in invaded hillslopes (Kakembo, 2004; Kakembo *et al.*, 2007; Manjoro *et al.*, 2012a). It was important to capture those different conditions in order to obtain results indicative of processes on the ground as well as to draw reasonable inferences regarding the hydrological impacts of the invader species. The experimental set-up was located on the same hillslopes to ensure uniformity in environmental conditions.

The hydrological impacts of the invader were quantified over both short- and medium-term temporal scales. Short-term included once off penetrometer and landscape organisation assessments. Medium-term monitoring was undertaken from September to November as the rainy season was targeted.

4.3.2 Landscape organisation assessment: Implication for surface hydrology

Landscape organisation assessment of invaded hillslopes was undertaken to investigate the degree of resource conservation, with specific interest to regulation of water resources, surface and subsurface flow paths mobilisation. The short term scale involved assessing soil surface conditions as a surrogate for infiltration. A hand-held penetrometer was used for this purpose and 100 readings were randomly taken from each subunit.

Furthermore, the Land Function Analysis (LFA) was conducted to assess the degree at which invaded landscapes facilitate the catchment's hydrological response. The Landscape Organisation Index (LOI) and associate indicators were important in this regard. The LFA was conducted in accordance to the LFA manual developed by Tongway and Hindley (2004).

4.3.3 Soil moisture monitoring

Soil moisture (cm^3/cm^3) was measured using the ECH2O EC-5 soil moisture probes of the METERGROUP, previously known as Decagon Group. Before installation of the soil sensors, sensor calibration was carried out.

Figure 4.11. On site soil sensor calibration and data retrieval

The probes were installed at a uniform depth of 15 cm below soil surface. Five soil moisture sensors were used. Considerations for soil moisture monitoring and installation of soil moisture probes included but were not limited to, installation of soil moisture sensors at a fairly uniform slope angle (7°) and slope position (middle slope) as well as uniform depth (ca. 15 cm). The soil moisture sensors measured soil moisture under the invader's tussock (1), in inter-patch areas (2), grass (3) and bare/eroded areas of the hillslope (4). The soil moisture content was used to indicate soil moisture retention capacity of the respective subunits; *viz* invaded, bare and grassland particularly after rainfall events. The findings were used to indicate the relative contributions of the respective subunits to subsurface flow and long-term inferences on hillslope hydrology within the catchment.

4.3.4 Surface runoff and sediment loss assessment

Gerlach troughs were used to quantitatively determine amount of surface runoff and sediment loss generated from hillslopes invaded by *P. incana*, bare-eroded and grass hillslopes (see Figure 4.12). The Gerlach troughs were placed at the base of the slope of each surface condition to capture surface runoff and sediments eroded from the hillslopes.

LAND USE/COVER TYPE	**GERLACH TROUGH ON SITE**
 Shrub invasion	
	

LAND USE/COVER TYPE	GERLACH TROUGH ON SITE
Grassland	
 Bare-eroded hillslope	

Figure 4.12. Pictorial illustration of land use/cover types and Gerlach Trough

Post rainfall measurements of the water contained by the Gerlach troughs were undertaken. The water inside Gerlach troughs was carefully emptied after individual rainfall events into a marked volume bucket and surface runoff was determined in litres (L). The recordings of surface runoff (L) were used as an indicator of runoff condition from the respective cover conditions. Data of the individual rainfall events were obtained from the climate station located at KwaNoncampa which is located at approximately 34 km away from the site.

Observations on accumulated sediments trapped by the Gerlach troughs were made and the sediments (g) were quantified after the monitoring period. The results of field sediment loss served to validate the SWAT hydrological model sediment loss outputs. This was achieved by comparing mean simulated sediment loss between September and November for 1994 and 2016 with hillslopes scale mean sediment loss.

4.4 Evaluating the influence of rainfall variability on the hydrology of the Keiskamma catchment

Responding to one of the research questions of the present study, the influence of rainfall on the hydrologic response of the catchment was also assessed. Daily rainfall and streamflow records were analysed using linear regression.

4.5 Statistical analysis of data

The Microsoft Office Excel data analysis package was used for statistical analysis of data and representation of results. Statistical tests were used to analyse data obtained in this study. Statistical tests are the most quantitative ways to determine whether hypotheses can be substantiated, or whether they must be modified or rejected outright (Hirsch *et al.*, 2002, page 97).

5 CHAPTER 5: RESULTS

Introduction

This chapter provides findings of this study. The results of the assessment of land use/cover change of the Keiskamma catchment responding to the first objective of the study are presented. First, classification accuracies of the classified land use/cover maps are presented. Second, classified land use/cover change maps of the Keiskamma catchment for 1994 and 2016 are shown. Quantitative land use/cover changes from 1994 to 2016 are also presented.

The second section addresses objective 2 of the study, and presents findings of the modelled hydrological response of the catchment simulated using the SWAT hydrologic model. First, this section presents model performance findings for the calibration and validation period. After this, hydrological responses of the catchment for 1994 and 2016 land use/cover scenarios are presented. The impacts of impoundments and rainfall variability on streamflow are also presented.

The third section presents results of the hydrologic impacts of *P. incana* in the catchment. Field experiment results of hydrological implications and impacts of *P. incana* invasion are presented. The findings include hydrologic implications of landscape organisation of invaded areas, surface conditions, surface runoff, soil moisture content as well as sediment loss in invaded areas, grassland and bare-eroded areas. This section also entails a validation of the SWAT model using the simulated and fieldwork sediment loss results.

5.1 Catchment land use/cover changes

5.1.1 Land use/cover maps accuracy

In the present study, an accuracy assessment of classified imagery was undertaken to assess the degree of reliability of the classified maps. Classification accuracy assessment achieved satisfactory overall accuracy of 87.2% and 87.12% for 1994 and 2016 classified maps respectively. Both maps had a Kappa Coefficient of 0.84 (see Table 5.1).

Table 5.1. Summary of accuracies (%) of classified land use/cover change maps for 1994 and 2016

Land use/cover classes	**1994**		**2016**	
	Producers'	**Users'**	**Producers'**	**Users'**
Built-up areas	90,40	97,70	80,91	100,00
Agricultural land	80,30	70,45	73,33	94,29
Dense vegetation	93,00	95,13	97,67	84,00
Rangeland	78,20	86,41	93,94	74,32
Bare and eroded land	97,30	85,27	84,83	94,97
Water	96,80	100,00	87,23	100,00
Overall accuracy	**87,20**		**87,12**	
Kappa accuracy	**0.84**		**0.84**	

5.1.2 Land use/cover change trend analysis

Findings of the assessment of land use/cover change obtained using supervised image classification show that the Keiskamma catchment had agriculture as the dominant land use activity in 1994 compared to the year 2016. In 1994, the central parts of the catchment appeared to be dominated by bare-eroded land whereas in 2016 the central parts were dominated by rangeland. In both years, dense indigenous forest as well as forest plantations dominated the upper watershed of the catchment and riparian vegetation was prominent along drainage lines towards the coastal areas (see Figure 5.1).

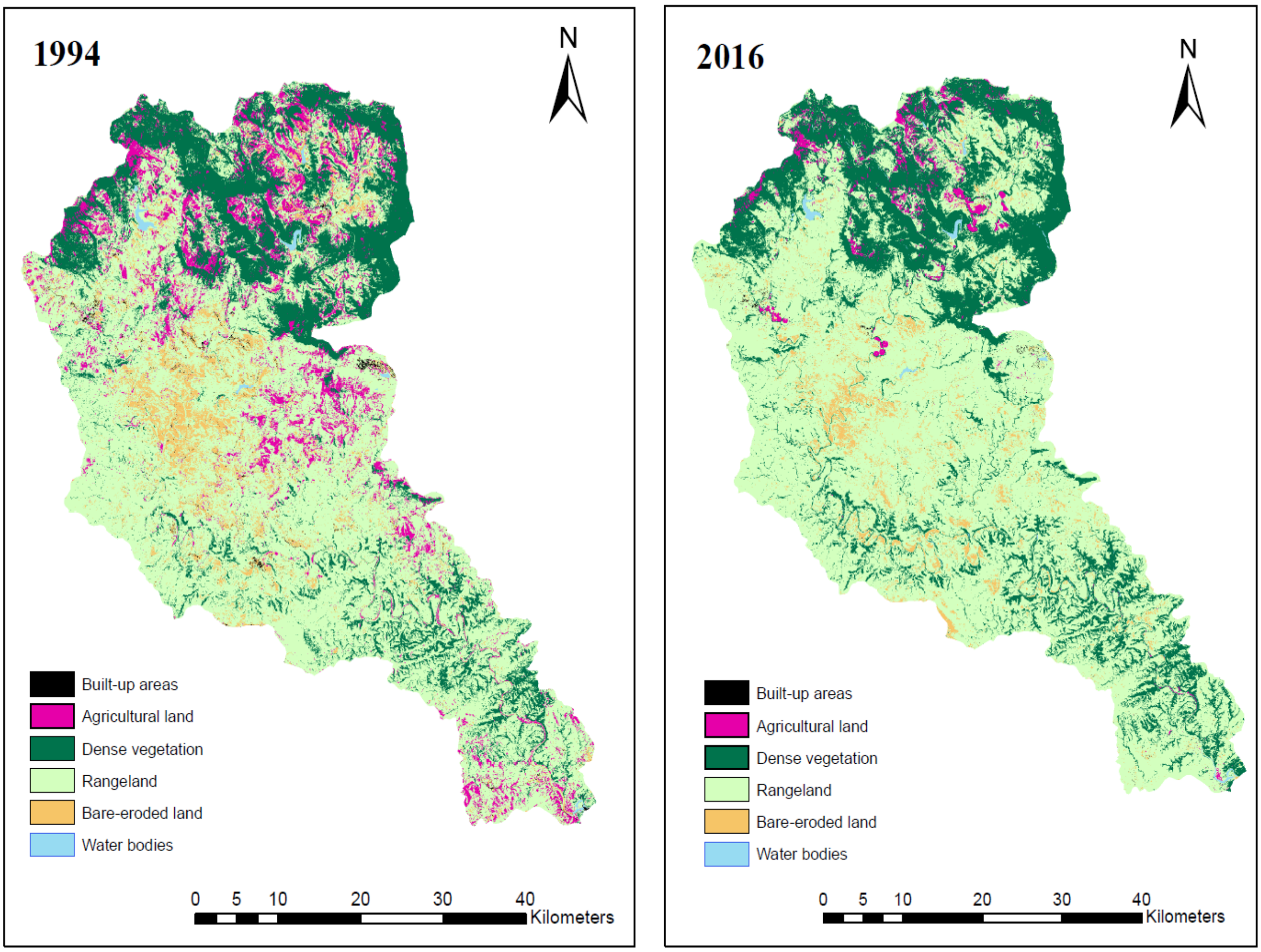

Figure 5.1. Land use/cover change maps for 1994 and 2016

The assessment of land use/cover changes in the Keiskamma catchment also indicates that rangeland and dense vegetation increased over the years. From 1994 to 2016, agricultural land and bare-eroded land decreased by 19386.63 ha and 6627.24 ha respectively (see Figure 5.2).

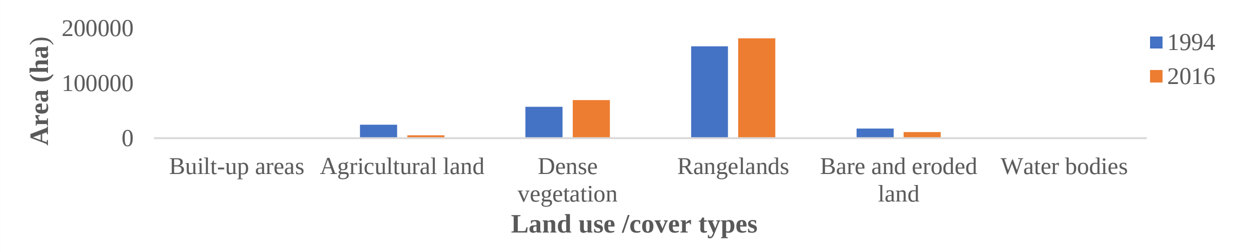

Figure 5.2. Land use/cover change for 1994 and 2016

In the assessment of land use/cover change, Rangelands shows the highest increase of 5.5 % whereas dense vegetation showed an increase of 4.5 %. Built-up areas, Agricultural and Bare-eroded land have decreased by 0.3 %, 7.1 % and 2.5 % respectively from 1994-2016 (see Table 5.2).

Table 5.2. Land use/cover change trend analysis from 1994 to 2016

land use/cover change Types	**Change from 1994-2016 (%)**	**Trend (Change)**
Built-up areas	-0.326	Decreased
Agricultural land	-7.194	Decreased
Dense vegetation	4.452	Increased
Rangelands	5.463	Increased
Bare and eroded land	-2.459	Decreased
Water bodies	0.064	Increased

5.2 Hydrological responses to land use/cover changes

5.2.1 Catchment scale hydrological responses to land use/cover changes

SWAT hydrologic model performance and uncertainty

Model performance of the hydrologic model used in the present study indicates that the SWAT model obtained a Good model performance score for the calibration period. NSE and R^2 were both 0.69, there was an RSR of 0.56 and PBIAS of -2.1. The p–factor was 0.77 and r-factor was 1.01. Mean observed and simulated streamflows were 3.51 m^3/s and 3.59 m^3/s respectively (Figure 5.3 **(a)**).

For the validation period the model performed unsatisfactorily. NSE and R^2 were 0.4, RSR was 0.79 and PBIAS was 0.3 However, very acceptable uncertainty indicators were obtained with a p-factor = 0.92 and r-factor of 1.38. Mean observed and simulated streamflows of 3.67 m^3/s and 3.66 m^3/s were obtained respectively (Figure 5.3 **(b)**).

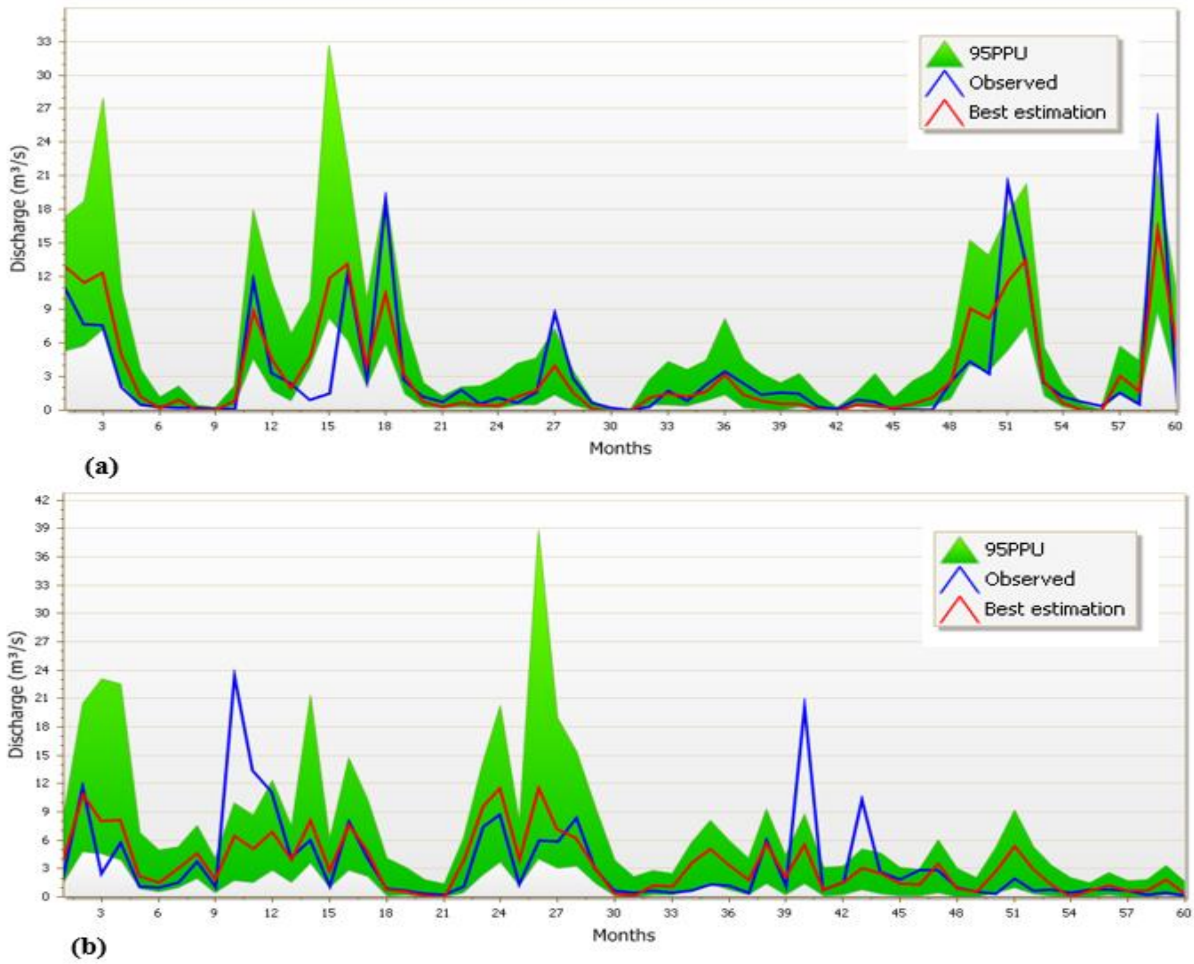

Figure 5.3. Observed and simulated monthly streamflow for (a) calibration (1996-2000) and (b) validation (2010-2016)

Hydrological responses to land use/cover changes

The hydrological response of the Keiskamma catchment modelled using the SWAT model shows that mean annual streamflow, surface runoff and sediment loss of the Keiskamma catchment decreased from 1994-2016. The streamflow of the Keiskamma catchment decreased significantly ($p<0.05$) by 71.43 % over the 22 year period. Surface runoff and sediment loss significantly decreased by approximately 98.79 % and 99.95 % respectively ($p<0.05$). A decrease in soil water content, evapotranspiration and groundwater was observed (see Table 5.3).

Table 5.3. Mean annual hydrological responses and change for land use/cover change scenarios of 1994 and 2016

Hydrological response	**Land use/cover change scenario**		**Change (%)**
	1994	**2016**	
Streamflow (m^3/s)	2.59	0.74	Decreased (71.43 %)
Surface runoff (mm)	0.39	0.004	Decreased (98.79 %)
Soil moisture (mm)	70.95	67.78	Decreased (4.45 %)
Evapotranspiration (mm)	48.59	46.00	Decreased (5.33 %)
Groundwater (mm)	16.49	3.38	Decreased (79.50 %)
Sediment loss (t/ha)	0.109	0.000059	Decreased (99.95 %)

Hydrological response to impoundments

The effect of impoundments located on the Keiskamma river, which was assessed by comparing simulated and gauged streamflow upstream and downstream of impoundments, indicates that the existence of dams in the Keiskamma catchment contribute to a reduction of downstream streamflow by 66.9 % of mean streamflow measured upstream of impoundments (see Figure 5.4). Mean discharge measured upstream and downstream of impoundments is 0.56 m^3/s and 0.18 m^3/s. There was a significant difference between upstream and downstream discharge ($p>0.05$).

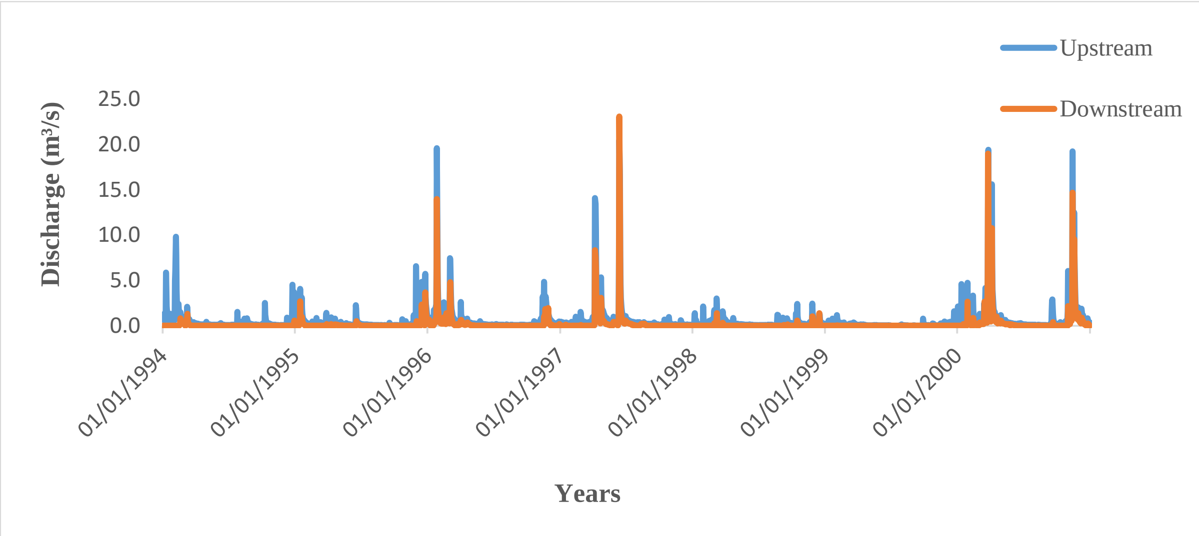

Figure 5.4. Effect of impoundments on streamflow

Overview of the impacts of land use/cover change on the hydrological response

To illustrate the influence of land use/cover changes on the hydrological response of the catchment, Table 5.4 provides an overview of the hydrological implications of land use/cover changes that occurred in the Keiskamma catchment over the last 22 years. An increase in dense vegetation (commercial forests and riparian vegetation) has implications for reductions of streamflow and surface runoff. Increases in rangelands (shrub) show the potential to decrease the hydrological response of the Keiskamma catchment.

Table 5.4. Overview of the impacts of land use/cover change on the hydrological response of the Keiskamma catchment

		INFLUENCE ON THE HYDROLOGICAL RESPONSE					
		Streamflow (m^3/s)	Surface runoff (mm)	Soil moisture (mm)	Evapotranspiration (mm)	Groundwater (mm)	Sediment loss (t/ha)
LAND USE AND LAND COVER CHANGE TREND	Built-up areas (**-0.32 %**)	-	-	-	-	-	-
	Agricultural land (**-7.19 %**)	↓	↓	↓	↓	↓	↑
	Dense vegetation (**+4.45 %**)	**↓**	**↓**	↑↓	↑	↑↓	**↓**
	Rangelands (**+5.46 %**)	↓	**↓**↑	**↓**	**↓**	↓	↑
	Bare and eroded land (**-2.45%**)	↓	**↓**↑	**↓**	**↓**	↓	↓
	Impoundments	**↓**	-	↓	-	-	-

DIRECTION OF INFLUENCE:

↑	Increase
↓	Decrease
↑	Strong increase
↓	Strong decrease
↑↓	Increase and or decrease
-	No influence

5.2.1 The influence of rainfall variability on the hydrology of the Keiskamma catchment

One of the study's research questions entailed assessing the influence of rainfall on the hydrology of the Keiskamma catchment through comparing rainfall records with observed streamflow. The results of the analysis indicate a strong correlation (R=0.71; R^2=0.56) between observed streamflow and rainfall for the calibration period from 1994 to 2000. There was a significant difference (p=0.033; $p<0.05$) between mean monthly streamflow and rainfall during the calibration period, whereas a weak correlation (R=0.43; R^2= 0.18) between streamflow and discharge existed for the validation period from 2010 to 2016 (see Figure 5.5).

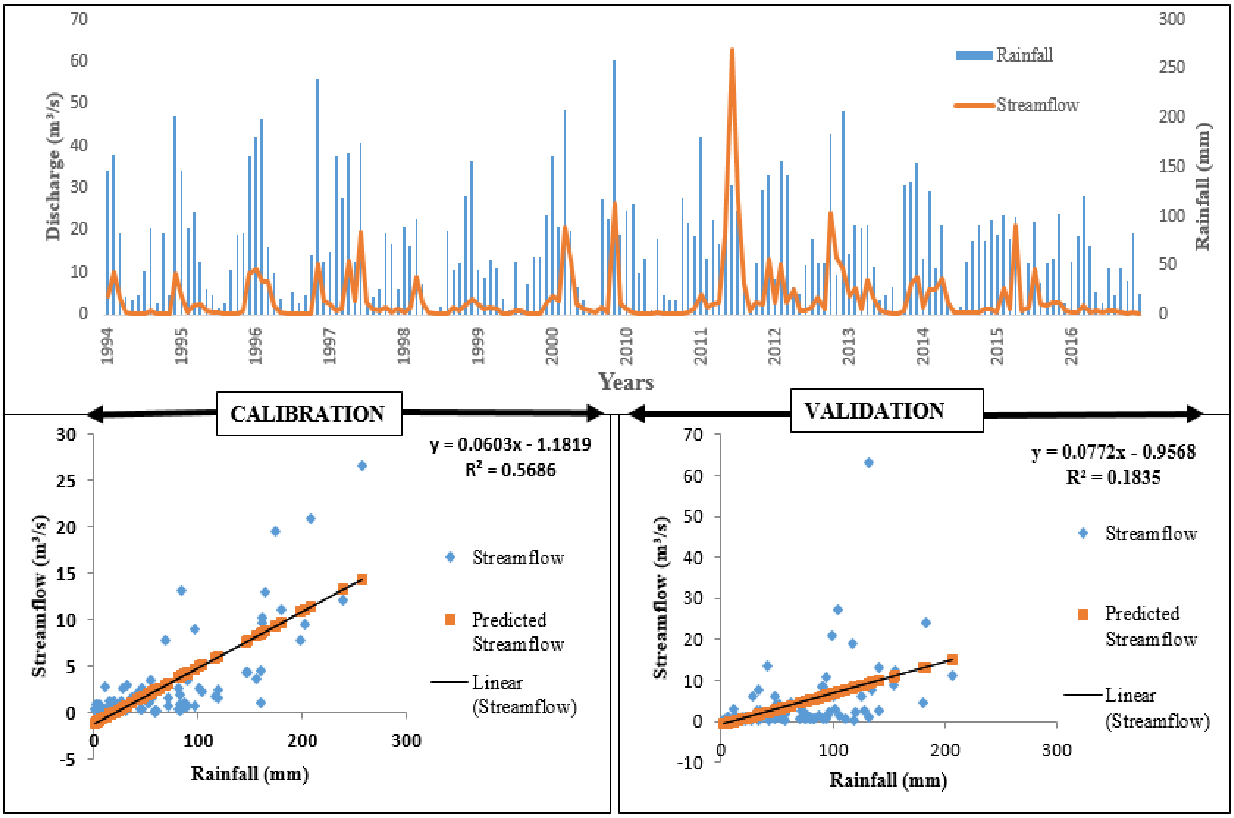

Figure 5.5. Relationship between catchment hydrologic response and rainfall for the calibration and validation period

5.2.2 Hydrological impacts of P. incana encroachment

Invaded hillslopes hydrological implications

To determine the hydrological implications of *P. incana* towards understanding its hydrological impacts, the LFA was used to assess degree of surface water regulation by *P. incana* invasion. Results of the LFA show that hillslopes invaded by *Pteronia incana* were characterised by large inter-patches, Bare-eroded areas, exposed rocks, thicket shrubs and *Aloes*. The **Landscape Organisation Index** (LOI) of the shrub invaded hillslopes was 0.4 which implies dysfunctionality of invaded hillslopes, particularly with large inter-patch eroded areas (see Table 5.5).

Table 5.5. Summary of Landscape Organisation data of *P. incana* shrub invaded hillslopes of the catchment

No. of patch zones per 10 m	**Total Patch zone width (m/10 m)**	**Average inter-patch length and range (m)**	**Landscape Organisation Index (LOI)**
5,8	1,24	4,44 (0.18 – 7.28)	0,4

Hydrological impacts of P. incana invasion

Soil surface conditions

Results for soil surface conditions under *P. incana,* bare-eroded areas and grassland which were obtained using a penetrometer are presented in this section. The results indicate that Bare-eroded areas and *P. incana* inter-patches had high mean surface hardness of 6.5 kg/cm² and 4.6 kg/cm² respectively. Grassland and the *P. incana* tussock (patch) have low surface hardness of 0.5 kg/cm² and 1.7 kg/cm² respectively (see Figure 5.6). The results for soil surface conditions have implications for infiltration such that *P. incana* inter-patch and eroded areas have low levels of infiltration, enhanced runoff generation and connectivity and ultimately erosion.

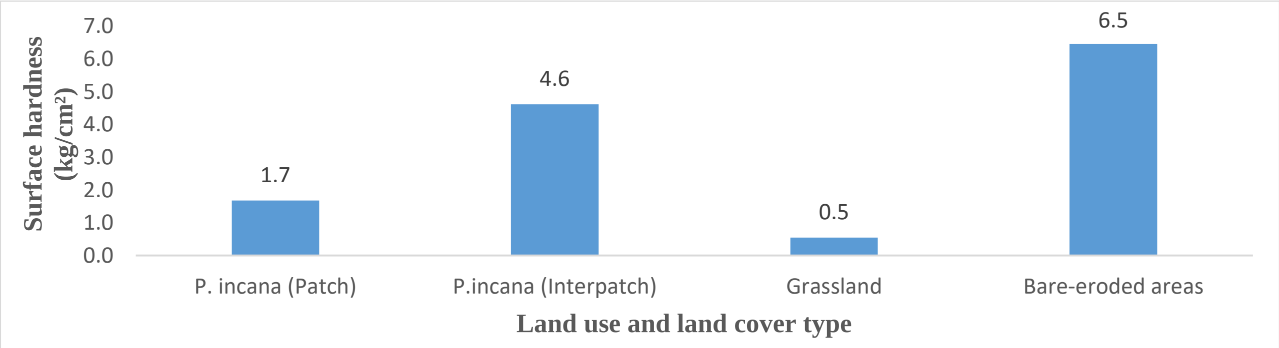

Figure 5.6. . Soil surface conditions of *P. incana* invaded areas, grassland and bare-eroded areas

Surface runoff after individual rainfall events

Surface runoff which was determined using Gerlach Troughs after individual rainfall events, shows that *P. incana* invaded and Bare-eroded areas had high surface runoff generation and low surface runoff was generated from grassland (see Figure 5.7). There was a significant difference between surface runoff generated under *P. incana* and grassland ($P<0.05$). There was no clear relationship between surface runoff and the individual rainfall events; however, there was a strong correlation ($r=0.72$) between rainfall the rainfall events and surface runoff generated under grassland.

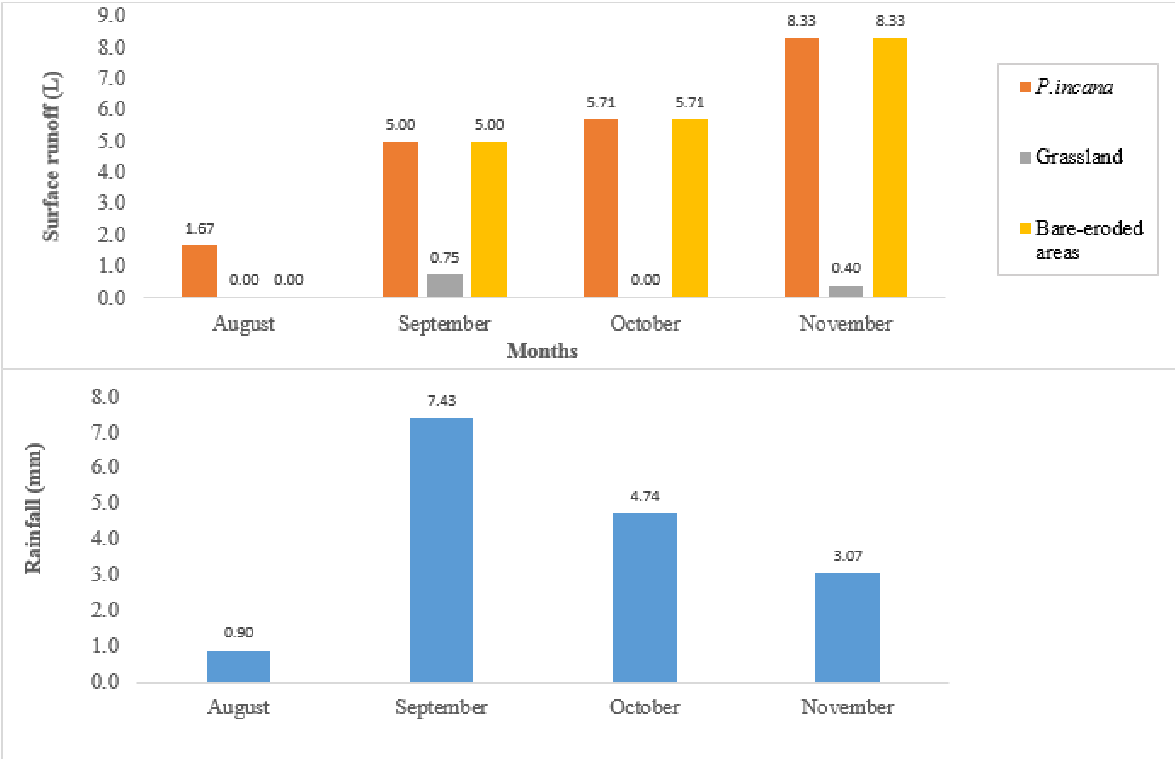

Figure 5.7. Surface runoff on *P. incana* invaded areas, grassland and bare-eroded areas and related rainfall events

Soil volumetric moisture content variations

As part of assessing hydrological impacts of *P. incana*, soil moisture content, particularly during and after rainfall events, was determined using soil moisture sensors. The results indicate that mean soil moisture was high under grassland (0.175 cm³/cm³) and *P. incana* (0.172 cm³/cm³), intermediate under *P. incana* inter-patches (0.124 cm³/cm³) and low in bare-eroded areas (0.099 cm³/cm³). There was no significant difference in soil moisture content between *P. incana* and grassland. A significant difference in soil moisture existed between the other land use/cover types and bare-eroded areas. For one high rainfall event which occurred on 06 September 2018, grassland (0.46 cm³/cm³) exhibited the highest soil moisture followed by *P. incana* at 0.36 cm³/cm³) (see Figure 5.8).

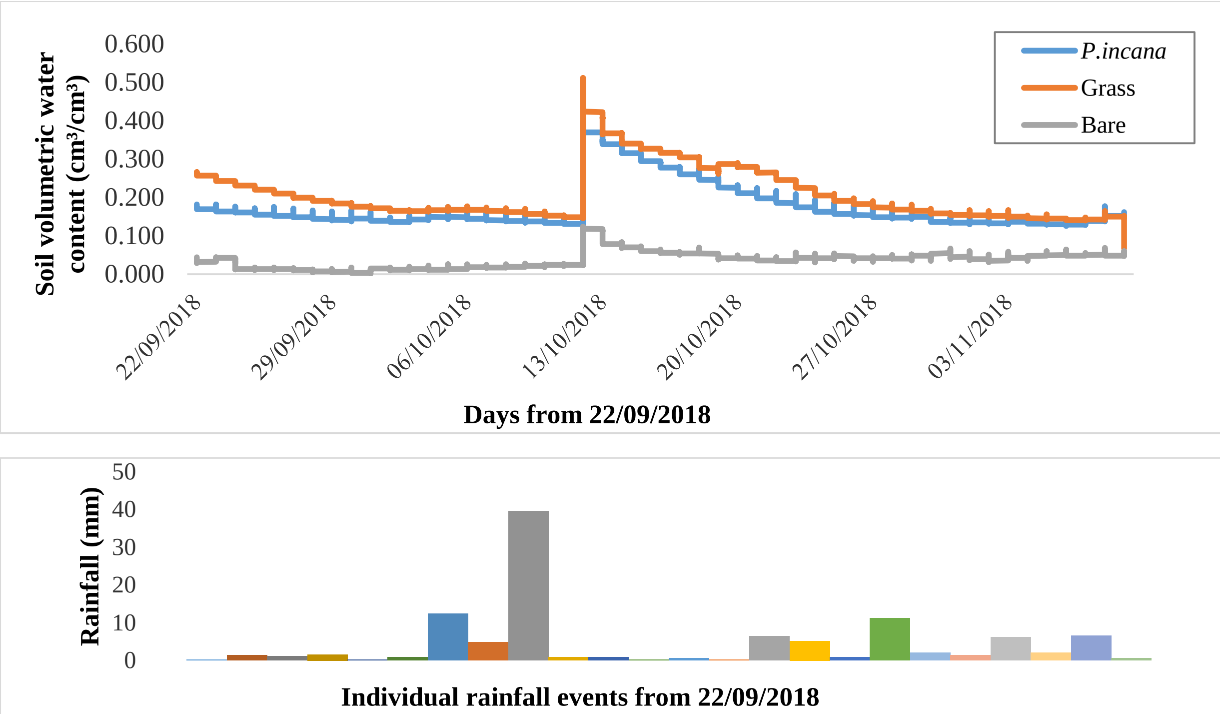

Figure 5.8. Soil moisture under *P. incana* invaded areas, grassland and bare-eroded areas in relation to rainfall events

Soil moisture content under the *P. incana* shrub showed to be higher compared to the shrub's inter-patch (see Figure 5.9). There was a significant difference between soil moisture content under *P. incana* patches and inter-patches. No significant difference was observed between soil moisture of *P. incana* patches and grassland. A significant difference between soil moisture conditions under *P. incana* patches and bare-eroded areas existed. There was no significant difference between *P. incana* inter-patches and grassland as well as inter-patches and bare eroded areas.

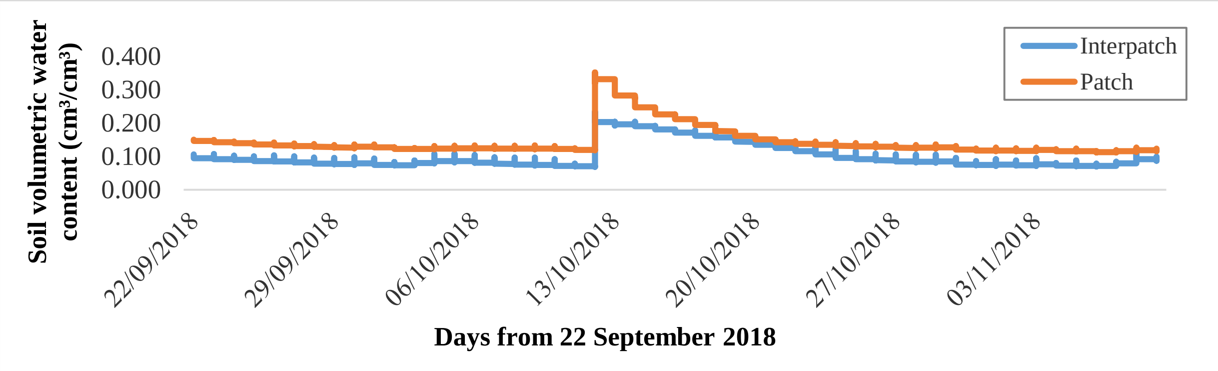

Figure 5.9. Soil moisture variations under *P. incana* patch and inter-patch

Sediment loss under P. incana, eroded-areas and grassland

As part of assessing hydrological impacts of *P. incana* shrub invasion, sediment loss was determined under *P. incana* shrub, eroded areas and grassland. Sediments accumulated in Gerlach Troughs were used to determine sediment loss under each land use/cover type. The results indicate that sediment loss is more profound in *P. incana* invaded areas (885 g) and bare-eroded areas (1056 g). Grassland hillslope showed no sediment loss (see Figure 5.10).

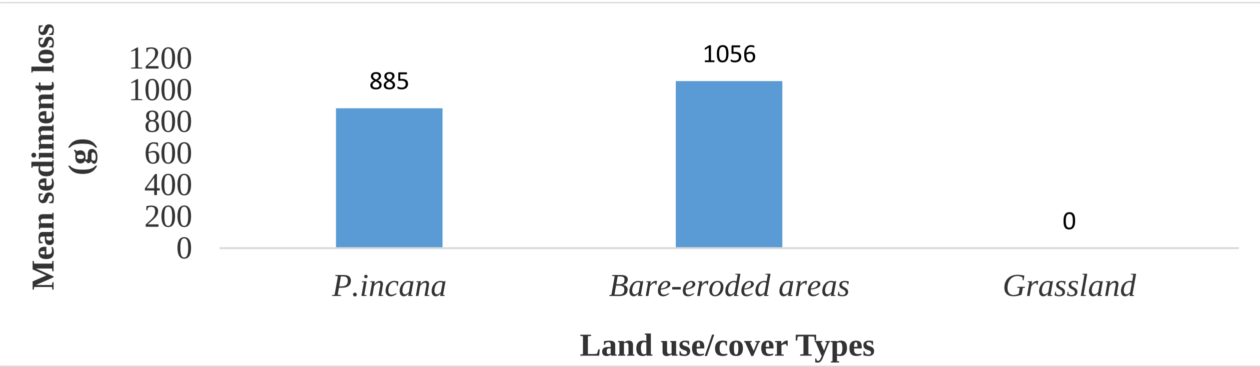

Figure 5.10. Sediment loss in *P. incana* invaded hillslopes, grassland and bare-eroded areas

Validating SWAT model simulations using fieldwork monitoring

Field monitoring of sediment loss served to validate the SWAT hydrological model sediment loss outputs. Catchment simulated sediment loss showed a decrease from 1994 to 2016 whereas hillslope sediment loss under *P. incana* and bare-eroded was high (see Figure 5.11).

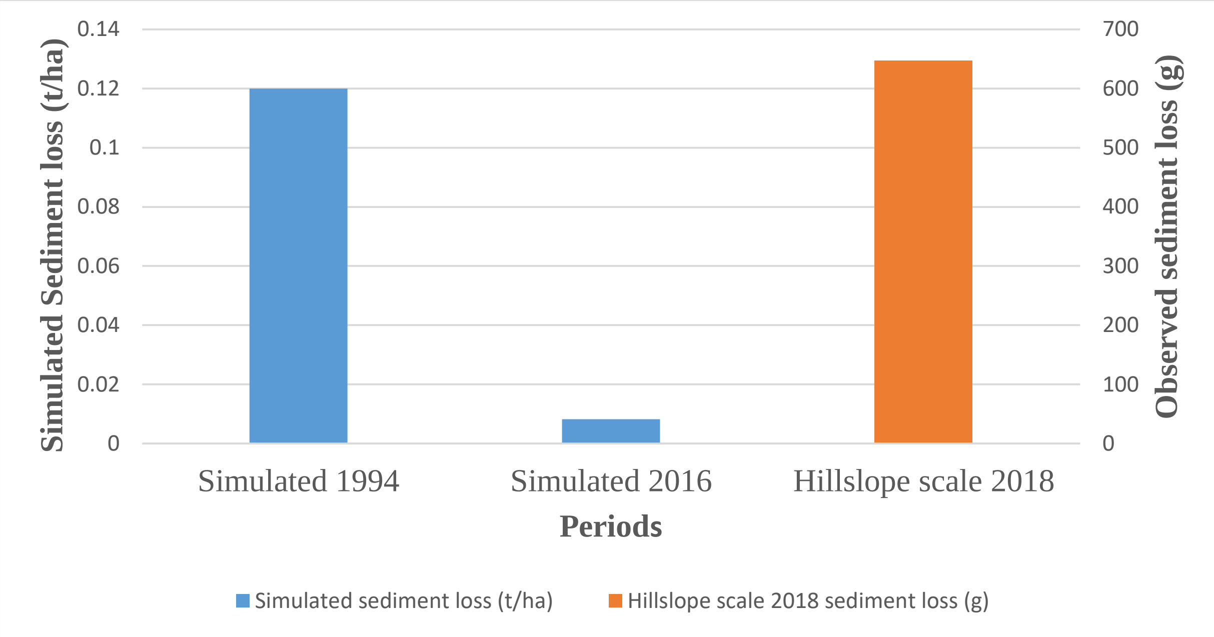

Figure 5.11. SWAT model validation using field monitored sediment loss

6 CHAPTER 6: DISCUSSION

Introduction

A discussion of the results, presented in the foregoing chapter, is provided in the following sub-sections; namely an assessment of land cover/use change of the Keiskamma catchment; the impacts of land use/cover changes on the hydrological responses of the catchment and then SWAT model performance. A discussion of the hydrological impacts of *P. incana* invasion onto the catchment is given. The influence of rainfall variability on the hydrological response of the catchment is also discussed. The last part of the discussion elucidates the overall critique of the SWAT model.

6.1 Keiskamma catchment land use/cover changes

The Keiskamma catchment has undergone considerable land use/cover changes owing to anthropogenic influences. The results of the study evidently show an increase in rangeland and a decline in bare-eroded areas and agricultural land. In addition, an increase in dense vegetation, constituted by indigenous and commercial forest and riparian vegetation is also noticeable. These land use/cover changes have a potential influence on the hydrological response of the Keiskamma catchment.

In contrast to the findings of Mhangara (2011), a decline of bare and eroded areas was observed. Secondary successions of mainly shrub species in bare-eroded and abandoned agricultural land explain the decline of bare-eroded areas. The establishment of shrub colonies in degraded and abandoned land also explains the mapped increase in rangelands in the Keiskamma catchment. Other studies, in similar catchments of the Eastern Cape also observe shrub encroachment proliferating from stream valley bottoms onto grassland and abandoned agricultural land (Kakembo and Rowntree, 2003; Kakembo, 2004; Manjoro *et al.*, 2012b). Notwithstanding the increase in shrub encroachment, the converse potentially exists for the grassland component of the catchment as shrubs tend to outcompete and replace grass in many catchments in the face of climate change and land use disturbances (Browning *et al.*, 2008; Caracciolo *et al.*, 2014). The phenomenon of shrub encroachment onto catchment areas is also a global ecohydrological issue and has impacts on the hydrological response of affected catchment areas (Silva *et al.*, 2001; Brook and Bowman, 2006; Sankaran *et al.*, 2008; Maestre *et al.*, 2009; Liu *et al.*, 2013).

According to Mhangara (2011), agricultural land of the Keiskamma catchment significantly declined from 1972 to 2006. Findings of the present study also show that agricultural land has declined by 7.2 % between 1994 and 2016, thus this trend is in line with the findings of Mhangara (2011). The drivers of declining agricultural land are manifold and include climate change, historical droughts, land reform policy history, land abandonment, lack of political will, lack of investment and other environmental, social and economic-politico factors (Kakembo and Rowntree, 2003; Kakembo, 2004; Haindongo, 2009; Mhangara, 2011; Zengeni *et al.*, 2016). These factors, or a combination of them, have proved to be persistent in the Keiskamma catchment as agricultural land has been decreasing since 1972.

Contrary to the finding of Mhangara (2011), an increase in dense vegetation (forests and riparian vegetation) was observed in the present study. The increase in dense vegetation in upper parts of the catchment is driven by an increase in forest plantations for commercial purposes. Uncontrolled sprawling of alien invasive stands of Black Wattle trees contribute to the growth of dense vegetation in the Keiskamma catchment. Black Wattle forest patches also occupy and confine into riparian zones throughout the catchment (Maphiri, 2009). The increase in forest vegetation has effects on the hydrologic response of the Keiskamma catchment, particularly in the upper watershed of the catchment. Increased riparian vegetation, particularly Black Wattle, has profound impacts on the instream ecohydrological process and interaction between rivers and the adjacent hillslope fluvial process (Wilcox *et al.*, 2017).

A decline in built-up areas was observed, and the change is underpinned by the ongoing rural decay which is driven by the influx of rural to urban migration (Satterthwaite *et al.*, 2010). Nonetheless, the influence of built-up areas on the hydrologic response of the Keiskamma catchment is localised in a few small spatial units of the catchment, thus, making the hydrologic impact of built-up areas insignificant and negligible. As aforementioned, the land use/cover changes prevalent in the Keiskamma catchment have implications for hydrological responses of the catchment. The sections below discuss the findings of the impacts of land use/cover change on the simulated hydrological responses of the catchment.

6.2 Hydrological responses to land use/cover changes

Land use/cover changes have ecological and hydrological impacts (Wang *et al.*, 2012b). This section discusses the modelled hydrological response of the Keiskamma catchment to the major land use/cover changes; *viz* increased rangeland (shrub encroachment), forests in the upper watershed and increased riparian vegetation. This is then followed by a discussion of the model's performance and its implications for decision-making when using the simulated results of the hydrological responses.

The hydrological response; *viz* streamflow, surface runoff, sediment loss, soil moisture, evapotranspiration and groundwater of the Keiskamma catchment, has shown a decline from 1994 to 2016. The streamflow of the Keiskamma catchment decreased significantly ($p<0.05$) by 71.43 % over the 22 year period. Surface runoff and sediment loss decreased by approximately 98.79 % and 92.66 % respectively. A decrease in soil water content and evapotranspiration by 4.45 % and 5.33 % respectively occurred in the catchment. The observed trends and their implications are discussed in the subsequent sections.

6.2.1 Hydrological response to increased forest plantations

Indigenous forests and commercial forest plantations are major vegetation types strongly associated with a reduction in streamflow, overland flow and soil water content (Cui *et al.*, 2012; Wang *et al.*, 2012c; Woldesenbet *et al.*, 2017). Findings of this investigation showed an increase in dense vegetation which is constituted by forests and these predominantly occupy the upper parts of the catchment.

Forest vegetation in the upper parts of the catchment contribute to streamflow reduction through reducing overland flow, soil recharge, evapotranspiration and water intake by roots (Cui *et al.*, 2012; Martins *et al.*, 2016). A wealth of earlier studies conducted in forested landscapes clearly demonstrate an inverse relationship between forest vegetation cover and streamflow (Bosch and Hewlett, 1982; Trimble *et al.*, 1987; Cui *et al.*, 2012; Honda and Durigan, 2016; Wang *et al.*, 2017).

Honda and Durigan (2016) observe that over a 5-year period, forest density increased annually by an average rate of 6.7 %, basal areas at 5.7% and rain interception by the canopies at 0.6 %

of the total rainfall. An increase in basal areas by 1 m^2/ha significantly reduced the amount of throughfall by 0.9 %, independent of the forest vegetation structure (Honda and Durigan, 2016). Therefore, increases in forest biomass contribute to reduction of surface runoff, soil recharge and streamflow contributions (Honda and Durigan, 2016).

The hydrological impacts of forests are exacerbated by the fact that forest plantation species include both indigenous and exotic trees. The impacts of exotic trees on streamflow reduction have been quantified and demonstrated by several studies (Van Wyk, 1986; Bosch 1979; Van Wyk, 1986; Le Maitre *et al.*, 1996; Rwizi, 2015; Le Maitre *et al.,* 2015). Bosch (1979) recorded an 82% reduction of streamflow over a period of 20 years in the Drakensberg, KwaZulu Natal following a secondary succession of pine trees in a formerly grassland catchment. Exotic trees are characteristic of high-water consumption traits associated with high transpiration, deep and widely extending rooting systems that maximise water use over natives (Calder and Dye, 2001). Moyo *et al.* (2009) state that *Acacia Mearnsii* (Black Wattle) significantly modifies hydrological responses through the high evapotranspiration rates which cause water loss and lower water tables. Thus, contributions of forests to streamflow reduction in the upper watershed of the catchment are driven by the coupling between indigenous and exotic forest tree species.

Furthermore, forests are associated with high soil moisture content and evapotranspiration (ET), which vary across spatial and temporal scales. Nonetheless, owing to the limited spatial distribution of forest vegetation cover in the catchment, their contribution to catchment scale soil moisture and ET budgets may not be significant. With the ongoing and expanding commercial forestry plantations, the hydrological impacts of forest can be envisaged as cumulative and exacerbated in the long-term, unless otherwise unforeseen circumstances halt the foresting activities in the catchment. The hydrologic impacts of forests in the upper parts of the catchment cascade to affect the downstream hydrological regime.

Conversely, the role of forests in water conservation, particularly in upper catchment parts cannot be underestimated and this can be optimally realised by halting the planting of exotic species and encouraging conservation of indigenous and other species that are suited for the local environment.

6.2.2 Hydrological response to impoundments and increased riparian vegetation

The existence of impoundments (dams) in the Keiskamma catchment contributes to the significant reduction of streamflow by 66.9 % of the mean streamflow measured upstream of the impoundments. Streamflow reduction which is also augmented by the upper catchment forest, is exacerbated by three dams; namely Mnyameni, Cata and Binfield, which also occupy the upper parts of the catchment, but downslope of the forested Amatola mountains. Sandile dam which is located in the transition between the upper and central parts of the catchment, as well as the centrally located Debe and Dimbaza dams, further contributes to streamflow reduction of the main Keiskamma River.

The impoundments in the Keiskamma catchment supply water to the surrounding rural areas and towns for domestic and agricultural purposes. Water abstraction from impoundments lowers volumes of the impoundments and reduces dam outflows (Schneider *et al.,* 2017). Intermittent discharge outflows from the impoundments also contribute to downstream streamflow reduction as hydrologic connectivity of upstream and downstream streamflow is dependent on the outflow from the impoundments (Heimann and Krempa, 2011) .

According to Phillips *et al.* (2005), the impact of the Livingston dam in Texas USA on streamflow reduction was not significant. However, sediment transport was significantly reduced. This implies that the degree of impacts by impoundments on flow regulation varies, although other impacts are universal. Other impacts of the impoundment included channel scour for about 60 km downstream as well as channel incision, widening, coarsening and channel slope decrease. Nonetheless, these impacts diminished with distance away from the impoundment (Phillips *et al.*, 2005).

Subsequent to the modification of fluvial processes, impoundments also have ecological impacts (Mathias and Batalla, 2005). Reservoir-induced reduction of streamflow variability and sediment sour create a niche for encroachment of invasive vegetation into the formerly active channel (Kondolf and Wilcock, 1996). Flow modification by impoundments disrupts ecological and hydrological conditions within which native riparian vegetation thrives and exotic species that were formerly excluded by such conditions gain a competitive advantage under the new altered ecohydrological regime (Mathias and Batalla, 2005). On the basis of observations by Reid and Moyle (2003), Mathias Kondolf and Batalla (2005) observed that in California, there are more exotic vegetation species in reaches downslope of dams than elsewhere. Thus, impoundments in the Keiskamma river have contributions on the observed increases in riparian vegetation. According to Maphiri (2009), Black Wattle forest patches also occupy and confine into riparian zones throughout the Keiskamma catchment. Therefore, this has also negatively impacted streamflow.

The findings of land use/cover change assessment of the present investigation revealed an increase in dense vegetation of the Keiskamma catchment, and riparian vegetation was a component of it. Several studies observe that riparian vegetation is likely to modulate surface and subsurface related processes only at local scale due to its limited spatial distribution in catchment areas (Pasch and Rouve, 1986; Bren, 1993; Piegay, 1997; Darby, 1999; Tabacchi *et al.*, 2000; Richardson *et al.*, 2007). Generally, riparian vegetation modified streamflow hydraulics through flow reduction, obstruction, diversion, friction and sediment trap (Tabacchi *et al.*, 2000). Water consumption of riparian vegetation underpins its contribution to streamflow reduction and this may be more profound in stream channels downslope of the impoundments in the Keiskamma catchment.

6.2.3 Catchment hydrological response to increased rangelands

The conversion of bare-eroded land to rangeland has implications for the hydrological response of the catchment. Specifically for the Keiskamma catchment, the decline in bare-eroded areas have implications for (1) reduction of surface runoff, sediment loss and streamflow (2) decreases in soil moisture and evapotranspiration and (3) reduced groundwater.

(1) Reduction of streamflow, surface runoff and sediment loss

Bare-eroded areas of the Keiskamma catchment are proliferated by revegetation by both native and exotic vegetation species. Invasive woody plant species such as *Acacia Karoo* and *Pteronia incana* are the notably prominent vegetation types colonising and thriving on disturbed areas of the Keiskamma catchment (Kakembo and Rowntree, 2003; Kakembo *et al.*, 2007; Kakembo, 2009). These vegetation types constitute the rangeland component of the catchment which increased over the investigation period. Rangelands of the catchment regulate surface and subsurface flow (Wilcox *et al.*, 2017). Vegetation succession holds direct implications for reduction of surface runoff, soil loss and ultimately overland flow contributions to streamflow (Li *et al.*, 2016). According to Bazan *et al.* (2013), wood plant encroachments also affect hydrological responses through augmenting both transpiration and interception which subsequently lead to lower streamflow.

According to Briske (2017), woody plant invasions affect streamflow and evapotranspiration in different ways, depending on the degree of aridity and seasonality, as well as the availability of subsurface water in catchment areas. The Keiskamma catchment has become predisposed to sporadic low rainfall regimes for both the typical wet and dry seasons and these are ideal conditions for shrub encroachment (D'Odorico *et al.*, 2012). Under such prolonged conditions, shrub encroachment reduces streamflow via exacerbating rangeland aridity and absorption of incident rainfall of short duration and low intensity, thereby reducing subsurface flow contributions to streamflow (D'Odorico *et al.*, 2012; Saintilan and Rogers, 2015).

(2) Soil moisture and evapotranspiration dynamics under woody plant encroachment

Encroachments of woody plants result in an increase in the fractional contribution of bare soil evaporation and evapotranspiration in catchment areas, hence promoting soil moisture loss (Scott *et al.*, 2006; Briske, 2017). Similarly, the increase in shrubland in the Keiskamma catchment has implications for reduction of soil moisture storage. Shrubs modify soil moisture through high water consumption traits and introduction of large inter-patches, thereby increasing soil exposure to oncoming radiation; hence, depleting soil moisture through evapotranspiration.

According to Honda and Durigan (2016), ET was lower in the non-vegetated site compared to vegetated sites owing to soil moisture content conservation differences. ET rates are dependent on soil water content conditions, soil water availability for plant uptake and plant physiology. Eldridge *et al.* (2015) explain that ET is mostly determined by the amount of soil moisture available for latent heat to transform liquid water into water vapour. Thus, the low soil moisture content of the catchment induced by shrub encroachment underpins the low ET conditions in the Keiskamma catchment.

(3) Groundwater response to land use/cover change

SWAT model results indicated that groundwater recharge also decreased by 79.50 % over the 22 year study period. The decline in groundwater of the Keiskamma catchment could be a positive feedback loop to climatic, ecological and anthropogenic pressures on groundwater. According to Peck and Williamson (1987), in higher rainfall catchments, groundwater recharge occurs seasonally and significantly. Conversely, medium rainfall catchments experienced no apparent groundwater recharge for several years (Peck and Williamson, 1987). Notwithstanding cognisance of the complex factors that determine groundwater, the Keiskamma catchment is also a low-medium rainfall catchment, and the decline in groundwater can be explained by the findings of Peck and Williamson (1987) as a likely occurrence of prolonged periods of no groundwater recharge.

Richardson *et al.* (2007) point out that woody phreatophytic plants, both trees and shrubs, native and or exotic, access groundwater for transpiration and thus consume more water than grassland, contributing towards groundwater decline. According to Scott *et al.* (2006), reduction of groundwater via vegetation consumption becomes prominent when phreatophytic, deep-rotted woody plants become abundant in catchment areas, as is in case of the upper-part of the Keiskamma catchment.

6.2.4 SWAT model performance in simulating the hydrological responses

In assessing the hydrological response of the Keiskamma catchment to land use/cover change, the model showed good and unsatisfactorily model performances for calibration (1994-2000) and validation (2010-2016) periods respectively. Model performance statistics (NSE and R^2 of 0.69; RSR of 0.56; and PBIAS of -2.1) indicate that a good model performance was obtained for calibration (Moriasi *et al.*, 2007). Nonetheless, the negative PBIAS indicates that the simulated streamflow exhibited a tendency of overestimating the observed streamflow (Moriasi *et al.*, 2007; Arnold *et al.*, 2012b). The p-factor index of 0.77 is a measure of the SWAT model's capability to capture uncertainties. According to Arnold *et al.* (2012), a p-factor of 1 implies that 100% of the observed data is within the 95PPU band and all the correct processes in the catchment are accounted for in the model (Moriasi *et al.*, 2007; Arnold *et al.*, 2012b; Qiao *et al.*, 2015). In the present study, 77% of uncertainties were captured by the SWAT hydrologic model. On the other hand, the r-factor which is the thickness of the 95PPU band, has an ideal value near 0, indicating high model performance. The combination of p-factor and r-factor obtained for the calibration period of this investigation together indicate acceptable model performance and uncertainty (Arnold *et al*., 2012b).

For the validation period the SWAT hydrologic model performed unsatisfactorily as NSE and R^2 were 0.4 and RSR of 0.79 (Moriasi *et al.*, 2007; Arnold *et al.*, 2012b). A very good PBIAS of 0.3 was an exception. Notwithstanding the unsatisfactory model performance, very acceptable uncertainty indicators were obtained as p-factor = 0.92 and r-factor of 1.38. From these model performance statistics, as well as the hydrographs of simulated versus observed streamflow, it can be deduced that the SWAT model's performance did not perform well in modelling hydrological responses to land use/cover change during validation. However, owing to the acceptable ranges of uncertainty indicators, the simulated hydrological responses can be used for decision-making but not with high confidence levels as the simulations of the calibration period.

6.3 Hydrological impacts of *Pteronia incana* invasion

Hydrological impacts of *P. incana* in invaded hillslopes of the Keiskamma catchment were investigated. As pointed out earlier, *Pteronia incana* is a non-palatable dwarf shrub associated with soil degradation which has extensively affected the Eastern Cape Province (Kakembo, 2004, 2009; Odindi, 2009; Kakembo *et al.*, 2012). The hydrological impacts of the invader shrub were compared with two land cover types; *viz* grassland and bare-eroded areas. Comparisons of hydrological responses of the three land use/cover types were of utmost importance to highlight pre- and post-land use/cover change conditions. Grassland hillslopes represented the pre-land use/cover change conditions whereas invaded and eroded hillslopes represented disturbed catchment conditions. This investigation was done within the landscape functionality analysis framework and its implications for hillslope and ultimately catchment hydrology. Hydrological contributions of invaded hillslopes to the overall catchment hydrologic response were of interest in order to validate simulated hydrologic response of the catchment.

6.3.1 Landscape functionality of invaded hillslopes and hydrological implications

Landscape functionality of invaded hillslopes was undertaken to investigate the degree of resource conservation, with specific interest in the implications for regulation of water resources, surface and subsurface flow paths mobilisation. The findings of this investigation indicated that *P. incana* invaded hillslopes are dysfunctional systems. The invaded hillslopes depicted severe soil erosion, aridity and inability to capture replacement materials for regeneration processes such as deposition, revegetation, soil formation and incident rainfall capture (Tongway and Hindley, 2004). The dysfunctional state of the invaded hillslopes was shown by the low Landscape Organisation Index (LOI) of 0. 4, which was quantified in the field. LOI >0.75 is recommended for functionality of rangelands (Tongway and Hindley, 2004). Observations in the field also indicated that the invaded hillslopes are characterised by soil surface sealing, crusting, lack organic matter accumulation and severe gully erosion. Landscape functionality components such as patch width (1.24 m) and average inter-patch length (4.44 m) indicated that invaded hillslopes are prominent run-off zones which enhance runoff connectivity. The large and bare inter-patches prevalent in invaded hillslopes favoured resource loss over resource accumulation through overland flow.

P. incana tends to exhibit a "Pseudo-complete vegetation cover." "Inferior vegetation species may create an impression of a healthy cover, yet they are highly ineffective in terms of erosion control" (Kakembo, 2004; page 2). The invader depicts a vegetation cover suggestive of soil surface protection from erosion forces whereas the invasion predisposes soil surfaces to disturbances (Kakembo, 2004; Kakembo *et al.*, 2012). This attribute of the invader is inconsistent with ecohydrological laws of vegetation cover, which are directly related to soil protection and infiltration (Wilcox *et al.*, 2017). Dense vegetation cover provides protection of the soil surface from the destructive impacts of rainfall and overland flow.

Based on the landscape functionality status of invaded hillslopes it can be deduced that they are characterised by pronounced surface runoff. The dysfunctional state of invaded hillslopes also contributes to subdued hydrological processes such as infiltration and seepage. Several studies also made similar observations on the relationship between *P. incana* invaded hillslopes and hydrological responses (Kakembo and Rowntree, 2003; Kakembo, 2004, 2009; Kakembo *et al.*, 2007, 2012; Odindi and Kakembo, 2011). Noteworthy is that at catchment scale, the SWAT model showed a decrease in overland flow and sediment loss which is not the case in invaded hillslopes of the Keiskamma catchment. However, scale underpins these contrasting findings. Catchment simulations lack detailed spatial resolution to reveal detailed hillslope processes.

6.3.2 Soil moisture dynamics of invaded and non-invaded hillslopes

Soil moisture variations under P. incana and bare-eroded areas

Findings of this investigation indicate that soil moisture content was low in *P. incana* invaded and bare-eroded hillslopes, whereas grassland retained a relatively high soil moisture budget. This trend is consistent with findings of a similar investigation conducted by Odindi and Kakembo (2011). The low soil moisture budget in invaded hillslopes can also be attributed to the dysfunctional status of the hillslopes. However, soil moisture dynamics under the shrub's tussock were higher compared to the shrub's inter-patches and bare-eroded areas. Variations in soil moisture can be attributed to several underpinnings including but not limited to vegetation, soil, slope, and land use characteristics. Bare-eroded hillslope are prone to greater soil moisture loss via excessive evaporation owing to lack of vegetative cover to intercept and absorb oncoming solar and thermal radiation (Wilcox and Thurow, 2014).

Likewise, the patchy distribution and large inter-patch length of the invader shrub subjects invaded hillslopes to low soil moisture retention through exposure to high evaporation, augmenting shrub transpiration and reducing infiltration. The root systems, organic matter entrapment, stemflow and shade-effect offered by the shrub's cover enables the tussock to hold relatively high soil moisture content compared to the inter-patches. According to Saintilan and Rogers (2015), the high soil moisture conditions under the tussock improve percolation of the shrub rooting system and allow it to assess groundwater as a survival strategy for droughts. Nonetheless, the function of the invader's tussock has a minor effect in promoting infiltration, organic matter accumulation and developing run-on zones since it covers a small total basal area extent on the invaded hillslopes. The hillslopes are dominated by exposed soil surface which promotes water loss and overland flow.

Soil surface conditions and hydrological impacts

The state of soil surface conditions plays an integral role in regulating the hydrological response of catchment areas. Field observations and assessments indicate that invaded and bare areas have high mean penetrometer readings from 4.6 to 6.5 kg/cm^3, indicative of low infiltration and percolation conditions. Eldridge *et al.* (2015) observed a high infiltration of 50 $mm.h^{-1}$ and 48.2 mm^{-1}under grasses and shrub tussocks respectively. Their findings empirically reveal the implications of the penetrometer readings for infiltration under the different land use/cover types.

On site observations also revealed lack of organic matter on the invaded and eroded areas which, amongst other factors, underlie the low soil moisture content under *P. incana* inter-patches and bare-eroded areas. It has been well documented that organic matter accumulation improves soil moisture holding capacity (Vaz *et al.*, 2001; Moran *et al.*, 2009; Canton *et al.*, 2011; Wang *et al.*, 2012c; Peng *et al.*, 2013). The reduced soil moisture content in invaded and eroded hillslopes has implications for subsurface flows. Owing to reduced hydraulic conductivity in these land cover types, reduction of subsurface flow contributions to groundwater is potentially prevalent.

Soil moisture dynamics under grassland

As aforementioned, grassland had the highest volumetric soil moisture content compared to *P. incana* and bare-eroded areas. Grass cover impedes overland flow, thereby reducing its velocity and allowing substantial opportunity for infiltration, which contributes to the high soil moisture content. Soil moisture holding capacity of grass cover has implications for subsurface flow contributions. According to Finca (2011), grassland and resident vegetation in watersheds possess the ability to function as a sponge that captures and retains rainfall during summer. A component of the captured and percolated rainfall is then released gradually during the dry season as baseflow into streamflow (Finca, 2011; Wang *et al.*, 2017).

Odindi (2009) indicated that *P. incana* invasions predominantly occur in isohyetic zones of 218 -487 mm mean annual precipitation. In the Keiskamma catchment, this isohyetic zone (487 mm) is prevalent in the dry central areas of the catchment where there is increased aridity owing to sporadic low rainfall, which subsequently reduces overland flow significantly in vegetated hillslopes of the catchment. Therefore, rainfall characteristics in the areas of encroachment amplify the impact of grass cover in reducing surface runoff.

6.3.3 Surface runoff and soil loss in invaded and non-invaded hillslopes

Surface runoff under P. incana and bare-eroded areas

Hydrological implications, including surface runoff conditions of *P. incana* invasion, have been mostly based on deductive explanations. This investigation quantitatively revealed that invaded and bare-eroded hillslopes areas generated greater surface runoff compared to grassland hillslopes. Small surface runoff was generated on grassland hillslopes. In the events of small rainfall amounts, it was observed that no surface runoff was generated in grassland areas. Invaded and bare-eroded hillslopes generated substantial amounts of surface runoff. The scanty distribution and large exposed inter-patches favoured generation of surface runoff in invaded hillslopes. The size of *P. incana's* tussock lacks the capability of obstructing overland flow (Kakembo, 2004). This reduces the ability of the invader to reduce the overland flow rate so as to promote seepage (Wilcox *et al.*, 2017).

Sediment loss in invaded and non-invaded hillslopes

As part of determining the hydrological impacts of *P. incana,* an assessment of sediment loss under shrub invasion, bare areas and grassland was made. The highest sediment accumulation was observed in Gerlach troughs placed on invaded and bare-eroded hillslopes. Conversely, very little sediment accumulation occurred in grassland hillslopes.

The replacement of grasses by invader shrubs exposes more of the soil surface to the action of raindrop impact, resulting in accelerated erosion and potential sedimentation, hence the high sediment loss in bare-eroded areas and *P. incana* covered areas (Abrahams *et al.*, 1994).

Earlier studies also observed that *P. incana* encroachment into catchment areas predisposes hillslopes to soil erosion and gullying (Kakembo and Rowntree, 2003; Kakembo *et al.*, 2007; Manjoro *et al.*, 2012a). The large inter-patch length between *P. incana* shrubs and the complete lack of vegetation cover in bare-eroded areas, promotes runoff connectivity, which favours erosive overland flow that weakens soil resistance, hence soil erosion. Moreover, *P. incana* encroachment characteristics do not solely underpin the high soil loss conditions in the Keiskamma catchment, but factors such as topographic controls, underlying problem soils and injudicious rangeland management approaches also contribute to soil loss (Kakembo *et al.*, 2007, 2009a; Mhangara, 2011; Nunes *et al.*, 2019).

According to Sun *et al.* (2017b), vegetation affords the soil protective cover against erosive incident rainfall and overland flow as it functions to bind soil particles into a firm cohesive state, thus reducing the predisposition of soil to erosion. In addition, the ability of grass to obstruct overland flow also reduces the overland flow erosive force, thus minimising soil's susceptibility to erosion. This explains the very low or no sediment loss on grass covered hillslopes.

6.4 Rainfall variability and hydrological response

The relationship between hydrological responses, particularly streamflow and rainfall, indicated a strong relationship for the calibration period and a weak relationship for the validation period. Nonetheless, the overall trend indicates a positive correlation between rainfall and streamflow. By implication, the hydrological response of the Keiskamma catchment for the validation period (1994-2000) was potentially influenced to a great extent by rainfall. The effect of rainfall on catchment hydrology is evident in the high hydrological response quantified for 1994 which corresponds with the high rainfall amounts of the respective year. Conversely, for the calibration (2010-2016) period, the influence of rainfall on the hydrological response of the catchment was subdued.

Land use/cover change conditions are likely to have dominated in determining the catchment's hydrological response. The sporadic and low rainfall events of the Keiskamma catchment have weakened the influence of rainfall on the hydrological response of the catchment in the last seven years. The decline in streamflow also reflects the significant decline is rainfall received in the catchment.

Odiyo *et al.* (2015) assessed long-term changes and variability in rainfall and streamflow in Luvuvhu River Catchment, South Africa. Their findings also showed a decline in rainfall corresponding with streamflow decline. Nonetheless, the authors also observed that streamflow decreases can also be induced by anthropogenic factors, such as impoundments and not necessarily and solely rainfall decrease (Odiyo *et al.*, 2015).

6.5 Critique of the SWAT model

As aforementioned, the SWAT model performed unsatisfactorily during the validation period. However, acceptable uncertainty indicators were quantified, which then compensated for poor model performance, thus qualifying model results. The poor model performance is multifaceted. The existence of substantial variations in catchment conditions for the calibration and validation period could explain this (USDA, 2004). Despite the poor model performance the SWAT model is also characteristic of certain limitations.

The SWAT hydrologic model is a highly complex modelling tool with a high number of parameters and extensive required input datasets. According to Arnold *et al.* (2012b), the complex nature of SWAT parametrisation is one of the model's inherent weaknesses. It is associated with complications in model parameterisation, calibration and validation processes. For catchment areas such as the Keiskamma, characterised by highly variable topography and climatic regimes, such attributes could exacerbate model complexity and hence the inherent weakness of the model.

Furthermore, rural catchment areas are mostly data scarce study sites for hydrological and climatic records. The limited availability and inconsistent quality of data interferes with the precision of SWAT model predictions.

Data gaps in observed streamflow and rainfall data have implications for uncertainty in the modelling process (Gassman *et al.*, 2007). The use of estimated data inputs for some parameters, rather than actual observed data, subjects the SWAT modelling process to uncertainties. The present study used the weather generator (WGEN) to download weather parameters such as humidity, solar radiation and wind speed, as records of observed data for a station in the catchment were unavailable. The use of estimated data to predict real processes is quite a limitation. Therefore, to improve hydrological modelling predictions, appropriate pre-planning and organising of climate and hydrological data should be at the forefront of future projects owing to data gaps in rural catchments, as these were also encountered in the present study.

The input catchment soil parameters used in the model were based on characteristics of soil mapped at global scale, the Harmonised World Soil Database, and this could have introduced substantial amounts of uncertainty in the modelling process. Similarly, the SWAT land use database parameters are based on American vegetation cover types (Gassman *et al.*, 2007; Arnold *et al.*, 2012b). Although this is generally accepted and suggested as adequate, it could still underlie some uncertainty during the modelling process of the present study.

Only one streamflow gauge was used as a source of observed streamflow for model calibration and validation. According to Abbaspour (2015), model calibration and validation using streamflow from more stations is ideal to adjust the SWAT hydrologic to local and regional watershed characteristics. This would enable better prospects for reliable model predictions of catchment hydrological response. Detailed information on spatial parameters is indispensable for building a correct watershed model. A combination of measured data and spatial analysis techniques using pedotransfer functions, geostatistical analysis, and remote sensing data would be the way forward (Abbaspour, 2015, page 6).

7 CHAPTER 7: CONCLUSIONS AND RECOMMENDATIONS

7.1 Conclusion

From the investigation in can be deduced that the major land use/cover changes that occurred in the Keiskamma catchment entailed an increase in woody vegetation encroachments into grassland, abandoned agricultural land and bare-eroded surfaces. Other notable land use/cover changes in the catchment included forest expansion in the upper parts of the catchment as well as a proliferation of invasive phreatophytic woody plants in the riparian zone, in response to streamflow regime modification by impoundments. By implication, land use/cover change of the Keiskamma catchment can be explained by a combination of ecological and anthropogenic drivers.

According to the SWAT model results, over the 22-year study period, the hydrological response; *viz* streamflow, surface runoff, sediment loss, soil moisture, evapotranspiration and groundwater of the Keiskamma catchment, has shown a decline. A significant decrease in streamflow, surface runoff and sediment by 71.43 %, 98.79 % and 92.66 % respectively was observed. Catchment soil water content and evapotranspiration decreased by 4.45 % and 5.33 % respectively. The modelled hydrological response can be used to inform catchment management decisions, owing to the satisfactory model performance and uncertainty levels of the simulations. Rainfall patterns also tie in closely to the modelled hydrological response of the Keiskamma catchment. Declines in rainfall amount, sporadic rainfall occurrence and drought conditions also could have influenced the hydrological response of the Keiskamma catchment over the past 22 years.

The observed land use/cover changes have intricate implications for the modelled hydrological responses of the catchment. The significant shrub encroachment has altered hillslope hydrological conditions, particularly by way of impeding and enhancing subsurface and surface flow respectively. This in turn underpins the observed catchment's hydrological response. Expansion of forests in the upper parts of the catchment, particularly exotic tree species has implications for reducing overland flow and subsequent contributions to streamflow.

Existence of impoundments and invasive vegetation encroachment into riparian zones of the catchment also amplify streamflow reduction in the Keiskamma catchment. So do the rainfall patterns. The assessed land cover/change explains the modelled hydrological response from different angles with each land use/cover change influencing different component(s) of the hydrological response.

Furthermore, from the present study, the hydrological impacts of *P. incana* shrub encroachment are revealed. The shrubs transform catchment hillslopes into run-off zones facilitated by the patchy distribution of the shrubs, characterised by large inter-patch areas as depicted by the low Landscape Organisation Index. Under *P. incana* inter-patches, usually bare-eroded areas, soil surface conditions are generally poor, characterised by soil crusting and surface sealing and have implications for reduced soil infiltration, as well as runoff generation and connectivity. Subsequent to the poor soil surface conditions under *P. incana* inter-patches and bare-eroded areas, low soil moisture and high surface runoff conditions were observed. Conversely, the tussocks of *P. incana* inherently have high soil moisture and surface conditions conducive to high infiltration. Grasses were noted to conserve catchment soil moisture, curtail overland flow and soil loss. Thus, shrub encroachment has negative effects on soil and water conservation in the catchment. The observed hydrological impacts of *P. incana* can be extrapolated to areas of similar hydrologic conditions. Moreover, cognisance of scale and scaling becomes imperative for understanding, predicting and extrapolating ecohydrological responses.

Field monitoring of sediment loss served to validate the SWAT hydrological model sediment loss outputs. Catchment simulated sediment loss showed a decrease from 1994 to 2016 whereas hillslope sediment loss under *P. incana* and bare-eroded areas was high. Similarly, hydrological responses such as overland flow were high at hillslope scale but low for the catchment. Therefore, the SWAT model results demonstrated a general sediment loss decline at catchment scale without depicting the variations at hillslope scale. The scale related differences in hydrological responses at hillslope and catchment level highlight the importance of scale considerations in understanding and predicting ecohydrological processes.

The investigation has produced baseline quantitative information about the hydrological impacts of land use/cover changes. Therefore, long-term ecohydrological and socio-economic implications and impacts of land use/cover change as well as the shrub invasion of catchments can be identified. With such information, catchment management bodies and communities are positioned to envision the future and develop informed remediation and adaptation plans.

7.2 Recommendations

Recommendations of this investigation ought to highlight and advise on interventions towards judicious catchment management as well future studies.

7.2.1 Predicting catchment scale management decisions

The SWAT hydrologic model has the potential to predict land use and climate change impacts on the catchment's hydrology. Thus, capabilities of such frameworks can be adopted for strategic planning and management of the catchment. However, aspects of unsatisfactory model performance can still be improved by taking into consideration local conditions and improving availability of data for the requisite model inputs.

7.2.2 Reviving ecohydrological integrity: Shrub encroachment management

Several previous studies warned that, inter alia, invasion of *P. incana* is an insidious driver of land degradation in affected catchments of the Eastern Cape (Kakembo and Rowntree, 2003; Kakembo, 2004; Kakembo *et al.*, 2007, 2009a, 2012; Odindi and Kakembo, 2011; Manjoro *et al.*, 2012b). Findings of this investigation also agree with the previous studies. This echoes the need for prompt interventions to curb *P. incana* invasion to rehabilitate the ecohydrological integrity of invaded areas. Proactive and sustainable rehabilitation approaches should be central to implementation of such initiatives.

Management strategies that advocate for conservation of grassland and minimising the degree of interspace disturbances by shrub encroachment must be considered towards reviving the ecohydrological integrity of the Keiskamma catchment, as these are likely to result in increased infiltration and landscape functionality (Eldridge *et al.*, 2015).

Invasive vegetation management plans should also be developed and implemented as part of improving the catchment's water yield. The programmes can focus on invasive vegetation eradication on both hillslopes and in riparian zones. Relevant environmental authorisation must be obtained from relevant authorities to ensure compliance of such initiatives.

7.2.3 Future research directions

Future studies, as facets of determining catchment hydrological responses can focus on complementing remote sensing, hydrological modelling process, in-depth climate change influence-implications as well integrating social perceptions on sustainable management of catchment areas.

To complement land use/cover change mapping, intensive groundthruthing as a means of assessing vegetation types reflecting on the remotely sensed data is imperative. Information about the actual type of woody encroachment (shrub versus trees) and the degree of encroachment in the catchment will be key to complement explanations of hydrological impacts of vegetation encroachment.

Hydrological modelling processes can be improved in many ways. According to Abbaspour (2015), uncertainty in the predictions from hydrological modelling is an issue of importance as catchment models are not immune to uncertainties. Therefore, hydrological modelling approaches ought to minimise uncertainty through ensuring that all processes and features of the catchment that could affect model outputs are taken into account (Abbaspour, 2015). Therefore, use of comprehensive observed data inputs and parameters into hydrologic models is essential.

Towards understanding future impacts of land use/cover change as well as climate change, future investigations can focus on scenario based hydrologic modelling to inform strategic water resource management planning. Community inclusion in the development of catchment management and climate change adaptations can also be assessed with a view to restoring the ecohydrological integrity of the Keiskamma catchment.

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9 APPENDICES

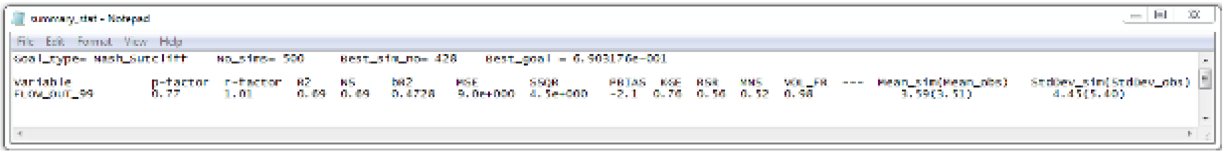
Appendix 1. Error Matrix: Accuracy assessment for 1994 land use map

Classified land use map	GCPS								
		Built-up areas	Agriculture	Dense vegetation	Rangeland	Bare-eroded land	Water	TOTAL	**Users Accuracy**
	Unknown	2	1	1	0	0	0	**4**	50.0
	Built-up areas	85	2	0	0	0	0	**87**	97.7
	Agriculture	0	155	6	59	0	0	**220**	70.5
	Dense vegetation	0	3	293	10	0	2	**308**	95.1
	Rangeland	1	19	15	248	3	1	**287**	86.4
	Bare and eroded land	6	13	0	0	110	0	**129**	85.3
	Water	0	0	0	0	0	92	**92**	100.0
	TOTAL	**94**	**193**	**315**	**317**	**113**	**95**	**1127**	
	Producers Accuracy	90.4	80.3	93.0	78.2	97.3	96.8		

Appendix 2. Error Matrix: Accuracy assessment for 2016 land use map

Classified land use map	GCPS								
		Built-up	Agricultural land	Dense vegetation	Rangeland	Bare-eroded land	Water	**Total**	**Users Accuracy**
	Unknown	0	1	0	0	0	0		
	Built-up areas	**89**	0	0	0	0	0	**89**	100.00
	Agricultural land	0	**99**	3	3	0	0	**105**	94.29
	Dense vegetation	0	12	**126**	10	0	2	**150**	84.00
	Rangeland	20	18	0	**217**	27	10	**292**	74.32
	Bare and eroded land	1	6	0	1	**151**	0	**159**	94.97
	Water	0	0	0	0	0	**82**	**82**	100.00
	Total	**110**	**135**	**129**	**231**	**178**	**94**	**877**	
	Producers Accuracy	80.91	73.33	97.67	93.94	84.83	87.23		

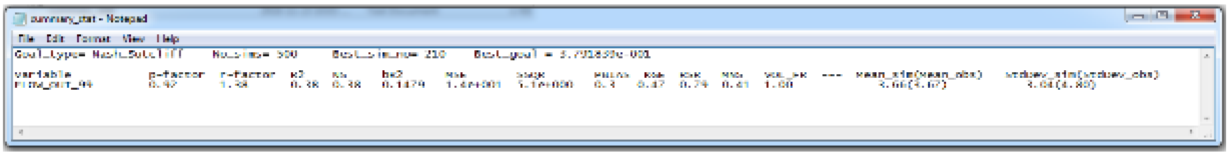
Appendix 3. SWAT model calibration model performance evaluation statistics



```
goal_type= Nash_Sutcliff    No_sims= 500    Best_sim_no= 428    Best_goal = 6.903176e-001

Variable      p-factor  r-factor  R2    NS    bR2     MSE       SSQR      PBIAS  KGE   RSR   MNS   VOL_FR  ---  Mean_sim(Mean_obs)  StdDev_sim(StdDev_obs)
FLOW_OUT_99   0.77      1.01      0.69  0.69  0.4728  9.0e+000  4.5e+000  -2.1   0.78  0.56  0.52  0.98         3.59(3.51)          4.43(5.40)
```

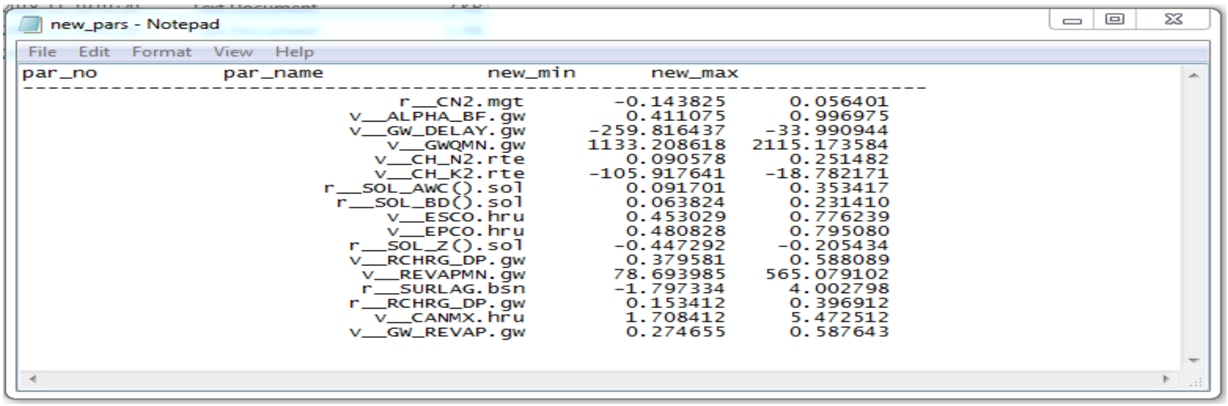
Appendix 4. SWAT model validation model performance evaluation statistics


summary_stat - Notepad

File Edit Format View Help

Goal_type= Nash_Sutcliff No_sims= 500 Best_sim_no= 210

Appendix 5. Best parameter values used for SWAT validation


new_pars - Notepad

File Edit Format View Help

par_no	par_name	new_min	new_max
	r__CN2.mgt	-0.143825	0.056401
	v__ALPHA_BF.gw	0.411075	0.996975
	v__GW_DELAY.gw	-259.816437	-33.990944
	v__GWQMN.gw	1133.208618	2115.173584
	v__CH_N2.rte	0.090578	0.251482
	v__CH_K2.rte	-105.917641	-18.782171
	r__SOL_AWC().sol	0.091701	0.353417
	r__SOL_BD().sol	0.063824	0.231410
	v__ESCO.hru	0.453029	0.776239
	v__EPCO.hru	0.480828	0.795080
	r__SOL_Z().sol	-0.447292	-0.205434
	v__RCHRG_DP.gw	0.379581	0.588089
	v__REVAPMN.gw	78.693985	565.079102
	r__SURLAG.bsn	-1.797334	4.002798
	r__RCHRG_DP.gw	0.153412	0.396912
	v__CANMX.hru	1.708412	5.472512
	v__GW_REVAP.gw	0.274655	0.587643

Appendix 6. t-Test: Two-Sample Assuming Equal Variances for mean monthly streamflow of 1994 and 2016

	Streamflow (1994)	*Streamflow (2016)*
Mean	2.596798598	0.739747238
Variance	13.70184384	0.214929643
Observations	12	12
Pooled Variance	6.95838674	
Hypothesized Mean Difference	0	
df	22	
t Stat	**1.724428767**	
P(T<=t) one-tail	0.049327491	
t Critical one-tail	**1.717144374**	
P(T<=t) two-tail	0.098654981	
t Critical two-tail	2.073873068	

Appendix 7. t-Test: Two-Sample Assuming Equal Variances for simulated soil moisture (mm) for 1994 and 2016.

	Year 1994	*Year 2016*
Mean	70.952	67.785
Variance	2031.886	173.761
Observations	12.000	12.000
Pooled Variance	1102.823	
Hypothesized Mean Difference	0.000	
df	22.000	

	Year 1994	Year 2016
t Stat	**-0.234**	
P(T<=t) one-tail	0.409	
t Critical one-tail	**1.717**	
P(T<=t) two-tail	0.817	
t Critical two-tail	2.074	

Appendix 8. t-Test: Two-Sample Assuming Equal Variances for simulated surface runoff (mm) for 1994 and 2016

	Year 1994	Year 2016
Mean	0.395	0.004
Variance	1.108	0.000
Observations	12.000	12.000
Pooled Variance	0.554	
Hypothesized Mean Difference	0.000	
df	22.000	
t Stat	**1.287**	
P(T<=t) one-tail	0.106	
t Critical one-tail	**1.717**	
P(T<=t) two-tail	0.211	
t Critical two-tail	2.074	

Appendix 9. Test: Two-Sample Assuming Equal Variances for simulated sediment loss (tons) for 1994 and 2016

	Year 1994	Year 2016
Mean	0.109999158	5.97643E-05
Variance	0.131004837	2.28633E-08
Observations	12	12
Pooled Variance	0.06550243	
Hypothesized Mean Difference	0	
df	22	
t Stat	**1.052204749**	
P(T<=t) one-tail	0.152063077	
t Critical one-tail	**1.717144374**	
P(T<=t) two-tail	0.304126154	
t Critical two-tail	2.073873068	

Appendix 10. t-Test: Two-Sample Assuming Equal Variances for simulated sediment loss of invaded hillslopes for 1994 and 2016

	SED_OUTtons (1994)	SED_OUTtons (2016)
Mean	0.368498341	0.008337117
Variance	1.239976639	7.91495E-05
Observations	12	12
Pooled Variance	0.620027894	
Hypothesized Mean Difference	0	
df	22	
t Stat	**1.120384174**	
P(T<=t) one-tail	0.137316156	
t Critical one-tail	**1.717144374**	
P(T<=t) two-tail	0.274632312	
t Critical two-tail	2.073873068	

Appendix 11. Regression statistics for streamflow and rainfall during calibration period

Regression Statistics	
Multiple R	0.75
R Square	0.56
Adjusted R Square	0.56
Standard Error	3.25
Observations	84

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	1145.67	1145.67	108.05	1.24E-16
Residual	82	869.38	10.60		
Total	83	2015.06			

	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*Lower 95.0%*	*Upper 95.0%*
Intercept	-1.18	0.54	-2.15	0.03	-2.27	-0.09	-2.27	-0.0935
Rainfall	0.06	0.005	10.39	1.243E-16	0.04	0.07	0.04	0.0746

Appendix 12. Regression statistics for streamflow and rainfall during validation period

Regression Statistics	
Multiple R	0.428353
R Square	0.183487
Adjusted R Square	0.173529
Standard Error	7.647314
Observations	84

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	1077.637	1077.637	18.42701	4.80624E-05
Residual	82	4795.475	58.48141		
Total	83	5873.113			

	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*	*Lower 95.0%*	*Upper 95.0%*
Intercept	-0.95682	1.511693	-0.63295	0.528529	-3.964060677	2.050415	-3.96406	2.050415
Rainfall	0.077202	0.017985	4.292669	4.81E-05	0.041424611	0.112978	0.041425	0.112978

Appendix 13. LFA field data for *P. incana* invasion Transect 1

Distance (m)	**Patch width (cm)**	**Patch/Inter-patch Identification**	**Inter-patch length (cm)**
		Bare with Gravel	50
0.52	18	Indigenous Shrub	
		Bare with Gravel	120
2.3	22.7	Patch Mosaic	
		Bare with Cracks	20
2.83	6.4	Indigenous Shrub (Purple)	
		Bare	235
3.2	16.6	*P. incana*	
4.26	133	Patch Mosaic	
5	15.3	Aloe Steam	
		Bare	76.4

Distance (m)	**Patch width (cm)**	**Patch/Inter-patch Identification**	**Inter-patch length (cm)**
5.9	9.3	*P. incana*	
		Bare	23
6.2	9.6	Succulent Plant	
		Bare with Gravel	90
7.3	17.3	Indigenous Shrub	
		Bare with Gravel	42
7.7	17	*P. incana*	
		Bare with Gravel	115
7.9	9	Indigenous Shrub	
		Bare with Gravel	90
8.8	11.7	Indigenous Shrub	
		Bare with Gravel	140
10	260	Giant Indigenous Shrub	
		Bare	91
12	13	*P. incana*	
		Bare	77
12.9	5.4	Indigenous Shrub	
		Bare with Gravel	41
13.3	6.3	Indigenous Shrub	
		Bare	90.6
14.3	4.3	Indigenous Shrub	
		Bare	141
15.7	3.3	Indigenous Shrub	
		Bare	13
16.2	11.3	*P. incana*	
		Bare	32
16.4	1.3	*P. incana*	
		Bare	53
17	9.3	Indigenous Shrub	
		Bare	170
20	15.6	Indigenous Shrub	
		Bare	190
24	13.4	Aloe	
		Bare	216
26.5	131	Patch Mosaic	
		Bare	55
27.5	13.8	*P. incana*	
		Bare	34
28.2	4.3	*P. incana*	
		Exposed Rock	34
30.6	13.7	*P. incana*	
		Exposed Rock	145
32.3	11.6	*P. incana*	
		Exposed Rock	280
36	14.4	*P. incana*	
		Bare	85
37.8	12.6	*P. incana*	
		Bare with Gravel	285
40.9	9.6	*P. incana*	
		Bare with Gravel	372
45	23.7	Patch Mosaic	
		Bare with Gravel	90
45.8	11	Aloe	
		Bare with Gravel	113
47	9.8	Aloe	

Distance (m)	Patch width (cm)	Patch/Inter-patch Identification	Inter-patch length (cm)
		Bare	184
49	27.3	*P. incana*	
		Bare	730

Appendix 14. LFA field data for *P. incana* invasion Transect 2

Distance (m)	Patch width (cm)	Patch/Inter-patch Identification	Inter-patch length (cm)
		Bare	180
2	4.7	Indigenous Shrub	
		Bare	110
14	6.4	Indigenous Shrub	
		Bare	120
15.5	23	*P. incana*	
		Bare	79
17	26.1	*P. incana*	
		Bare	30
17.5	5.7	*P. incana*	
		Bare	170
18.1	5.3	*P. incana*	
		Bare	127
21.4	7.2	*P. incana*	
		Bare	61.3
22.1	3.7	Patch Mosaic	
		Bare	120
23.5	23.4	*P. incana*	
		Bare	37
24.3	10	Aloe	
		Bare	50
24.4	13	*P. incana*	
		Bare	80
25.5	12.4	*P. incana*	
		Bare with Gravel	150
27.2	23.4	Patch Mosaic	
		Bare	110
29	12	*P. incana*	
		Bare	130
30.4	10	Grass Tussock	
		Bare	150
32	11	*P. incana*	
		Exposed Rock	330
35.5	32	Patch Mosaic	
		Exposed Rock	144
37.3	12	*P. incana*	
		Bare with Gravel	470
42.3	15.3	*P. incana*	
		Bare	190
44.7	13	Aloe	
		Bare	37
46.2	18.6	*P. incana*	
		Bare	70
47	8	Aloe	
		Bare with Gravel	80
48.3	2.4	*P. incana*	
		Bare	70

Appendix 15. LFA field data for P. incana invasion Transect 3

Distance (cm)	Patch width (cm)	Patch/Inter-patch Identification	Inter-patch length (cm)
	2.5	Sward	
		Bare with Gravel	48.6
3	25.4	*P. incana*	
		Bare with Gravel	110
4.4	58.7	Patch Mosaic	
		Bare	140
7.4	7	*P. incana*	
		Bare	66
9	27.7	*P. incana*	
		Bare	80
11.1	11	Rock	
		Exposed Rock	310
14.1	8.7	*P. incana*	
		Bare with Gravel	51
14.7	21	*P. incana*	
		Bare	83
15.8	16	*P. incana*	
		Bare	14
16.4	9.4	*P. incana*	
		Exposed Rock	81
18.3	31	*P. incana*	
		Exposed Rock	91
19.7	2.7	*P. incana*	
		Bare	30
20.1	4.3	*P. incana*	
		Bare	148
21.6	13.8	*P. incana*	
		Bare	60
22.5	4	Indigenous Shrub	
		Bare	250
25	23	*P. incana*	
		Bare	60
26	57	Giant Shrub	
		Bare	147
27.5	9	*P. incana*	
		Bare	201
31.8	8	Indigenous Shrub	
		Bare	230
32.3	8	*P. incana*	
		Bare	75.4
33	56.4	*P. incana*	
		Bare	70
34.4	57.4	*P. incana*	
		Bare	92
36	36	*P. incana*	
		Bare	56
37	44.7	*P. incana*	
		Bare	74
38.3	13.8	Indigenous Shrub	
		Bare	180
40.4	11	*P. incana*	
		Bare	260
43.4	14.4	*P. incana*	

Distance (cm)	Patch width (cm)	Patch/Inter-patch Identification	Inter-patch length (cm)
		Bare	97
44.3	17.5	*P. incana*	
		Bare	447
50	13.4	*P. incana*	

Appendix 16. LFA field data for P. incana invasion Transect 4

Distance (m)	Patch width (cm)	Patch/Inter-patch Identification	Inter-patch length (cm)
		Bare	1.3
1.4	3.7	*P. incana*	
		Bare	12.6
14	5.2	*P. incana*	
		Bare	1.5
15.5	17.4	*P. incana*	
		Bare	1.5
17	20.1	*P. incana*	
		Bare	0.5
17.5	4.7	*P. incana*	
		Bare	0.6
18.1	5.3	*P. incana*	
		Bare	3.3
21.4	6	*P. incana*	
		Bare	0.7
22.1	5.8	Patch Mosaic	
		Bare	1.4
23.5	19	*P. incana*	
		Bare	0.8
24.3	9	Aloe	
		Bare	0.1
24.4	12.7	*P. incana*	
		Bare	1.1
25.5	11	*P. incana*	
		Bare with Gravel	1.7
27.2	24	Patch Mosaic	
		Bare	1.8
29	11.7	*P. incana*	
		Bare	1.4
30.4	11	Grass Tussock	
		Bare	1.6
32	10.6	*P. incana*	
		Exposed Rock	3.5
35.5	34	Patch Mosaic	
		Exposed Rock	1.8
37.3	10	*P. incana*	
		Bare with Gravel	5
42.3	13.6	*P. incana*	
		Bare	2.4
44.7	12	Aloe	
		Bare	1.5
46.2	18.3	*P. incana*	
		Bare	0.8
47	10.9	Aloe	
		Bare	1.3

Distance (m)	Patch width (cm)	Patch/Inter-patch Identification	Inter-patch length (cm)
48.3	2.7	*P. incana*	
		Bare	1.7

Appendix 17. LFA field data for *P. incana* invasion Transect 5

Distance (m)	Patch width (cm)	Patch/Inter-patch Identification	Inter-patch length (m)
0.5	300	Sward	
0.52	12.7	*P. incana*	
		Bare	1.48
2	23.6	Patch Mosaic	
		Bare	0.6
2.6	6.4	*P. incana*	
		Bare	1
3.6	15	*P. incana*	
		Bare	1.7
5.3	14.1	Aloe	
		Bare	0.7
6	9.3	*P. incana*	
		Bare	0.2
6.2	10	*P. incana*	
		Bare with Gravel	1.3
7.5	21	Patch Mosaic	
		Bare with Gravel	0.2
7.7	17	*P. incana*	
		Bare with Gravel	0.2
7.9	9	*P. incana*	
		Bare with Gravel	0.9
8.8	10.3	Indigenous Shrub	
		Bare with Gravel	1.2
10	260	Patch Mosaic	
		Bare	2
12	13	*P. incana*	
		Bare	1.1
13.1	4.9	*P. incana*	
		Bare with Gravel	0.5
13.6	6.3	*P. incana*	
		Bare	0.7
14.3	5.6	*P. incana*	
		Bare	1.4
15.7	5	*P. incana*	
		Bare	0.5
16.2	10.3	*P. incana*	
		Bare	0.2
16.4	2.6	*P. incana*	
		Bare	5.3
17	9.3	*P. incana*	
		Bare	3
20	14.7	Indigenous Shrub	
		Bare	4
24	13	Aloe	
		Bare	2.5
26.5	140	Patch Mosaic	
		Bare	1

Distance (m)	Patch width (cm)	Patch/Inter-patch Identification	Inter-patch length (m)
27.5	12	*P. incana*	
		Bare	1.8
28.2	5.7	*P. incana*	
		Exposed Rock	1.8
30	13	*P. incana*	
		Exposed Rock	2.6
32.6	10.6	*P. incana*	
		Exposed Rock	3.4
36	13.2	*P. incana*	
		Bare	1
37	12	*P. incana*	
		Bare with Gravel	3.9
40.9	8	*P. incana*	
		Bare with Gravel	4.1
45	25	Indigenous Shrub	
		Bare with Gravel	0.3
45.3	10	Aloe	
		Bare with Gravel	1.4
46.7	9	*P. incana*	
		Bare	2.2
48.9	13	*P. incana*	
		Bare	7.3

Appendix 18. t-Test: Two-Sample Assuming Equal Variances for soil moisture between P. incana patch and inter-patch

	P. incana (Patch)	*P. incana (Inter-patch)*
Mean	0.172348173	0.122346312
Variance	0.006162803	0.003798334
Observations	6128	6128
Hypothesized Mean Difference	0	
df	11600	
t Stat	**39.21851095**	
P(T<=t) one-tail	0.00000	
t Critical one-tail	**1.644984997**	
P(T<=t) two-tail	0	
t Critical two-tail	1.960168512	

Appendix 19. Test: Two-Sample Assuming Equal Variances for soil moisture between P. incana patch and grassland

	P. incana (Patch)	*Grassland*
Mean	0.172348173	0.173180279
Variance	0.006162803	0.014135014
Observations	6128	6128
Pooled Variance	0.010148908	
Hypothesized Mean Difference	0	
df	12254	
t Stat	**-0.457207961**	
P(T<=t) one-tail	0.323764841	
t Critical one-tail	**1.644977985**	
P(T<=t) two-tail	0.647529682	
t Critical two-tail	1.960157595	

Appendix 20. Test: Two-Sample Assuming Equal Variances for soil moisture between P. incana patch and bare-eroded areas

	P. incana (Patch)	*Bare-eroded areas*
Mean	0.172348173	0.096136682
Variance	0.006162803	0.02655181
Observations	6128	6128
Pooled Variance	0.016357307	
Hypothesized Mean Difference	0	
df	12254	
t Stat	**32.98442662**	
P(T<=t) one-tail	5.7022E-229	
t Critical one-tail	**1.644977985**	
P(T<=t) two-tail	1.14E-228	
t Critical two-tail	1.960157595	

Appendix 21. Test: Two-Sample Assuming Equal Variances for soil moisture between P. incana inter-patch and grassland

	P. incana (Inter-patch)	*Grassland*
Mean	0.122346312	0.173180279
Variance	0.003798334	0.014135014
Observations	6128	6128
Pooled Variance	0.008966674	
Hypothesized Mean Difference	0	
df	12254	
t Stat	**-29.71547691**	
P(T<=t) one-tail	9.8385E-188	
t Critical one-tail	**1.644977985**	
P(T<=t) two-tail	1.9677E-187	
t Critical two-tail	1.960157595	

Appendix 22. Test: Two-Sample Assuming Equal Variances for soil moisture between P. incana inter-patch and bare-eroded areas

	P. incana (Inter-patch)	*Bare-eroded areas*
Mean	0.122346312	0.096136682
Variance	0.003798334	0.02655181
Observations	6128	6128
Pooled Variance	0.015175072	
Hypothesized Mean Difference	0	
df	12254	
t Stat	11.77714153	
P(T<=t) one-tail	3.80062E-32	
t Critical one-tail	1.644977985	
P(T<=t) two-tail	7.60124E-32	
t Critical two-tail	1.960157595	

Appendix 23. Test: Two-Sample Assuming Equal Variances for soil moisture between grassland and bare-eroded areas

	Grassland	*Bare-eroded areas*
Mean	0.173180279	0.096136682
Variance	0.014135014	0.02655181
Observations	6128	6128

Pooled Variance	0.020343412
Hypothesized Mean Difference	0
df	12254
t Stat	29.8998518
P(T<=t) one-tail	5.821E-190
t Critical one-tail	1.644977985
P(T<=t) two-tail	1.1642E-189
t Critical two-tail	1.960157595

Appendix 24 Monthly simulated sediment loss for 1994.

RCH	YEAR	MON	AREAkm2	SED_INtons	SED_OUTtons
84	1994	1	14.07	0.1075	0.1075
84	1994	2	14.07	3.896	3.896
84	1994	3	14.07	0.01848	0.01848
84	1994	4	14.07	0.01063	0.01063
84	1994	5	14.07	0.0119	0.0119
84	1994	6	14.07	0.000001196	0.000001196
84	1994	7	14.07	0.02096	0.02096
84	1994	8	14.07	0.0001089	0.0001089
84	1994	9	14.07	0.01639	0.01639
84	1994	10	14.07	0.03	0.03
84	**1994**	**11**	**14.07**	**0.2746**	**0.2746**
84	1994	12	14.07	0.03541	0.03541

Appendix 25. Monthly simulated sediment loss for 2016.

RCH	YEAR	MON	AREAkm2	SED_INtons	SED_OUTtons
84	2016	1	14.07	0.005896	0.005896
84	2016	2	14.07	0.02069	0.02069
84	2016	3	14.07	0.02673	0.02673
84	2016	4	14.07	0.01288	0.01288
84	2016	5	14.07	0.002301	0.002301
84	2016	6	14.07	0.0009428	0.0009428
84	2016	7	14.07	0.001773	0.001773
84	2016	8	14.07	0.0009346	0.0009346
84	2016	9	14.07	0.004269	0.004269
84	2016	10	14.07	0.002659	0.002659
84	**2016**	**11**	**14.07**	**0.01777**	**0.01777**
84	2016	12	14.07	0.0032	0.0032