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HYDROLOGICAL IMPACTS OF SUSTAINABLE LAND MANAGEMENT IN THE OLIFANTS SUB-BASIN

Neliswa Pretty MTHETHWA

A dissertation submitted to the Faculty of Engineering and the Built Environment, University of the Witwatersrand, in fulfilment of the requirements for the degree of Master of Science in Engineering.

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HYDROLOGICAL IMPACTS OF SUSTAINABLE LAND MANAGEMENT IN THE OLIFANTS SUB-BASIN

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DECLARATION

I declare that this research report is my own unaided work. It is being submitted for the Degree of Master of Science to the University of the Witwatersrand, Johannesburg. It has not been submitted before for any degree or examination in any other University.

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18 day of October (year) 2021

ABSTRACT

Sustainable Land Management (SLM) is the most effective tool in addressing different forms of land degradation. Despite this, the adoption of SLM practices and technologies remains low. Particularly due to the lack of an enabling environment, reliable and regulated impact monitoring systems/studies and incentives measures to encourage SLM adoption in different land-use activities. This study supports water resources managers, policymakers and land users to adopt SLM practices and technologies. As for water resources management, it is vital to adopt a system that integrates water and land resources, since any land use practice is, effectively, a decision with significant implications for water resources. Furthermore, it is essential to understand the impacts of management practices on water resources before implementation. To increase the potential success rate and help water resources managers make informed decisions, hydrological models, such as the Soil and Water Assessment Tool (SWAT), allow for quantitative assessment of these impacts.

The SWAT model is an essential tool that has been used worldwide to address water questions and help water practitioners make informed decisions. With the aid of the SWAT model, the study assessed the impacts of SLM practices and technologies on streamflow and sediment yield of the Olifants sub-basin. Running the SWAT model requires a substantial number of datasets from climatic data to spatial information of soil and land use. The necessary climatic, physiographic and hydrological datasets were collected from various sources and collated accordingly. Average monthly streamflow and sediment data were then used to calibrate and validate the model. The impacts of the selected practices and technologies were simulated by optimizing appropriate model parameters.

The SWAT model was able to simulate streamflow and sediment yield satisfactorily. The coefficient of determination (r^2) and Nash-Sutcliffe (NS) had values greater than 0.5. Root mean square error (RSR) values were all less than 0.7 and per cent bias (PBIAS) had values below 25 per cent, indicating satisfactory model performance. All the SLM practices used in this study reduced surface runoff and sediment yield. Filter strips had no impacts on the streamflow but decreased sediment yield. The introduction of agroforestry resulted in a decrease in sediment yield and total water yield. The decrease in net water yield is attributed to increased actual evapotranspiration due to increased tree cover. Contour farming is accountable for a 34% and 46% decrease in streamflow and sediment yield respectively. Sediment yield and streamflow were reduced by 87% and 20% respectively,

as a result of parallel terraces simulation. Adopting these practices in the sub-basins will prove advantageous.

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LIST OF ABBREVIATIONS

ALPHA_BF	Baseflow alpha-factor
ALPHA_BNK	Base flow alpha factor for bank storage
CANMX	Maximum canopy storage
CFRS	Climate forecast reanalysis system.
CH_COV2	Channel cover factor
CH_EROD	Channel erodibility factor
CH_K2	Effective hydraulic conductivity in the main alluvium
CH_N2	Manning's "n" value for the main channel
CN/CN2	Curve Number
DWS	Department of Water and Sanitation
ESCO	Evaporation compensation factor
FAO	Food and Agriculture Organisation
GW_DELAY	Groundwater delay
GW_REVAP	Groundwater "revap" coefficient
GWQMN	Threshold depth of water in the shallow aquifer required for return flow to occur
HRU_SLP	Average slope steepness
M	Million
NS	Nash-Sutcliffe
PBIAS	Per cent Bias
PRF_BSN	Peak rate adjustment factor for sediment routing in the main channel.
R^2	Coefficient of determination
RCHRG_DP	Deep aquifer percolation fraction
REVAPMN	Threshold depth of water in the shallow aquifer for "revap" to occur
RSR	Root mean square-observation standardisation
SAWS	South African Water Services
SIM	Small-holder Irrigation Management
SLM	Sustainable Land Management
SLSUBBSN	Average slope length
SOIL_AWC	Soil available water capacity (SOIL_AWC)
SOL_BD	Soil Density
SOL_K	Saturated hydraulic conductivity
SPCON	Channel sediment routing

SPEXP	Exponent parameter for calculating the sediment re-entered in channel sediment routing
SURLAG	Surface runoff lag time
SWAT	Soil and Water Assessment Tool
USLE_C	USLE C factor applicable to land cover/plant
USLE_K	Soil erodibility factor in USLE
USLE_P	USLE equation support parameter

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1. INTRODUCTION

1.1. Preamble

Sustainable land management (SLM) is at the forefront of addressing environmental concerns and rehabilitation. It is a champion in the fight against the long-standing and ongoing worldwide concern known as land degradation. Land degradation is a long-term loss of ecosystems, goods, and services, negatively affecting billions of people's livelihoods and food security (Nkonya *et al.*, 2016). Many livelihoods depend on land resources such as water for subsistence and income generation.

Therefore, it is crucial to understand the interlinkages of water and land resources on mesoscale such as sub-basins. Sub-basins play a significant role in providing ecosystem goods and services such as freshwater, regulation of floods, and retention of sediments. The degradation of such areas hinders their ability to provide the needed ecosystem services that improve water quality and the quality of life for inhabiting communities.

Factors such as topography, climate, chemical and physical soil properties contribute to land degradation. These direct causes of land degradation are well understood, and significant work has been done to understand their causal mechanisms (Mirzabaev *et al.*, 2016). For example, steeper slopes are more susceptible to water-induced soil erosion. Soils with high salinity content are naturally prone to degradation (Bonilla and Johnson, 2012). The causes and drivers of land degradation are complex and interrelated, with often context-dependent characteristics (Pender *et al.*, 2009; Nkonya *et al.*, 2011; von Braun *et al.*, 2013).

Increasing human population pressure and altering land management practices lead to continuous cultivation and exploitation of land resources. These human activities contribute to land degradation and unsustainable agricultural intensification (Jayne *et al.*, 2014). Agricultural and farming activities such as ploughing, heavy machinery, and overgrazing alter the physical landscape. These activities cause soil erosion, which subsequently pollutes surface water bodies through siltation (Issaka and Ashraf, 2017). Any land use practice is, effectively, a decision with significant implications for water resources. Changes in land surface affect rainfall's partitioning into infiltration, evapotranspiration, and runoff, consequently influencing the quantity and quality of water that flows or gets stored within a river system.

Furthermore, changes in catchment yields, dissolved organic carbon, and nutrient transfers are related to land management practices (Weatherhead and Howden, 2009). Low or absence of vegetation cover and poor soil physicochemical properties decrease the amount of water infiltrating into the soil, thereby increasing overland flow that is laden with sediments and nutrients, particularly during heavy rains or storms. The runoff pollutes nearby water bodies which, when used for water supply, lead to increased water treatment costs and reduced storage capacity due to siltation (Le Roux and Sumner, 2012; Pimentel and Burgess, 2013; Seutloali *et al*., 2017).

Land degradation has severe impacts on the environment and human livelihoods through interrelated, indirect, and complex processes. It reduces soil infiltration capacity, leading to higher surface runoff and declining dry season flows because of reduced subsurface storage of water during wet seasons (Qazi *et al*., 2017). Degraded catchments have more pronounced seasonal flow extremes and rapid drying up of streamflow compared to well-forested catchments (Qazi *et al*., 2017). Vegetation acts like a detention pond that absorbs and stores wet season rainfall and slowly releases the water during the subsequent dry season (Qazi *et al*., 2017).

The degradation of land resources is more than an environmental problem. It is a major global issue with severe impacts on soil productivity, water availability and threatens food security. Addressing land degradation should be approached in an all-inclusive manner, mixing different facets of sustainable management such as the environment and socio-economic elements. Attempts have been made to address this ongoing concern through technologies (e.g., World Overview of Conservation Approaches and Technologies and Land Degradation Assessment in Drylands), conventions (UNCCD), policies, and national programmes (e.g., LandCare). However, these initiatives are essential, but they fail to reverse, let alone halt land degradation completely.

Therefore, there is a need for adopting a more inclusive, holistic, and participatory land management approach as encapsulated in SLM. This type of land management is inclusive because it encourages multilevel, multi-stakeholder involvement, and partnerships at all levels (land users, technical experts, and policymakers). SLM relies on the principle of integrating socio-economic principles with environmental concerns to maintain and enhance productivity while reducing the level of production risk (Motavalli *et al*., 2013). SLM is restorative as it aims to reduce and restore degraded ecosystems through different sustainable land practices and technologies. SLM is land user-driven and promotes a participatory approach. Different approaches can be used to implement SLM.

Some of the most widely used approaches are participatory rural appraisal and farmer field schools (Gurtner *et al*., 2011).

Empirical evidence indicates that if widely adopted, SLM practices can prevent, reduce, and revert land degradation (Schwilch *et al*., 2014; Mwangi *et al*., 2015; Sanz *et al*., 2017). There is increasing scientific evidence of the potential benefits of adopting SLM measures as land-based solutions to address land degradation holistically (Gurtner *et al*., 2011; Motavalli *et al*., 2013; Mwangi *et al*., 2015; Sanz *et al*., 2017; FAO, 2017b). SLM approach also achieves other co-benefits, such as protecting biodiversity and securing the quantity and quality of soil and water.

SLM projects and programmes across different scales are becoming more favoured due to their ability to engage multiple sectors and stakeholders (Bunning *et al*., 2016). SLM calls for attention at all levels and across sectors to improve resource use and minimise conflicts over resources. At the same time, it demonstrates the benefits of interlinking food security, poverty, livelihood, land, and water governance concerns (FAO, 2017a). It encourages the integration of social, economic, physical, and biological values to achieve a more productive and healthier ecosystem (Motavalli *et al*., 2013).

Despite the ongoing and increasing impacts of land degradation on ecosystem services and increasing SLM practices, investments are unsatisfactory, especially in developing countries (Nkonya *et al*., 2016). The slow adoption is partly due to the SLM targets and indicators being mostly site- and nation-specific (Kust *et al*., 2017). The low investment in SLM results from the lack of coherent and evidence-based policy frameworks. Other key factors that affect the adoption and upscaling of SLM practices include the absence of incentive measures, reliable and regulated monitoring systems, and capacity building. SLM practices and technologies need to be defined based on the local context, informed planning, and data collection to increase knowledge before interventions. The participatory planning process enhances community awareness and involvement. Stakeholders who are well-informed about the benefits of adopting SLM practices and have been involved in the planning process are more likely to implement such practices.

There are five SLM groups referred to as cropping, cross-slope barriers, water, grazing, and forest management. Croplands and mixed land-use systems (crop-tree and crop-grazing) house most cropping management practices. These practices involve the use of organic and inorganic plant nutrients, minimum disturbance of the Soil, crop rotation, contour farming, and permanent soil cover. Crop rotation involves growing different types of crops in the same area during different growing seasons. Contour farming is a form of

sustainable cropping management in which ploughing, planting, and cultivating are done along the slope rather than downslope (Mwangi *et al*., 2015). Permanent soil cover means leaving cover on the field all year round.

These technologies improve soil fertility and indirectly benefit water resources through increased infiltration and reduced surface runoff. The main advantages are onsite conservation of rainwater by reducing surface runoff, evaporation loss and soil erosion by up to 50 per cent, thereby achieving fewer tillage operations, and reducing costs (Schwilch *et al*., 2014). Contour farming increases infiltration and reduces soil erosion and sedimentation, thus improving water quality (Arabi *et al*., 2008; Stevens *et al*., 2009).

Water management includes rainwater harvesting, grassed waterways, improved irrigation efficiency, and provision of drinking water for domestic and livestock use. Other water harvesting technologies include harnessing water from an upstream catchment area and directly diverting it to cropping fields. This process increases water availability. Water management is involved with different land uses, mostly related to crop production through irrigation and water harvesting, and water supply systems. The main aim of water management technologies is to improve soil-water storage, crop–water productivity, and integrated soil–crop–water management (FAO, 2017a).

Cross-slope barriers are remedial measures applied on steep lands in the form of earth or soil bunds, stone lines, barriers in gullies, and vegetative strips to reduce the slope's length. These cross-slope technologies also reduce runoff velocity and soil loss, thereby contributing to soil and water conservation. Terraces are usually constructed and developed over time behind earth bunds, vegetative filter strips (i.e., grass), or stone barriers, due to soil movement from the upper to the lower part of the terrace (Oladele and Braimohi, 2014). Terraces effectively reduce diffuse pollution from agricultural lands, especially on steep slopes (Arabi *et al*., 2008; Santhi *et al*., 2006). They are applied to various land-use systems but mostly in croplands.

Grazing land management practices include timing of rotation, controlled grazing, and resting periods. Controlled grazing refers to the intentional degree of control or management level applied to rangeland through grazing systems such as rotational stocking, continuous stocking, and strip grazing (White and Wolf, 2009). This technique is based on the principle of leaving the rangeland protected (by excluding grazing for 2 to 3 years, this period may vary) to allow the plant cover to recover. Controlled grazing results in increased forage harvested by animals, improved forage quality, extended grazing seasons, reduced fertiliser and herbicide applications, and reduced labour and feed costs

(Zaied, 2011). Rangeland resting, adjusted for seasonal variation, increases feed intake, reduces degradation risk, and protects vegetation resulting in increased income for farmers (Alkemade *et al*., 2013).

Forest management includes afforestation, assisted regeneration of forests, fire control, and other indirect practices such as improved cooking stoves. The alternate use of cooking stoves protects naturally-seeded tree seedlings. It encourages biodiversity through beekeeping and sustainable harvesting of non-wood forest products (FAO, 2017b). For instance, one thousand one hundred and forty-five (1 145) cooking stoves were distributed to communities in different catchments across the transboundary Kagera Basin. Such practice reduces deforestation, mitigates climate change, supports livelihood, and boosts soil fertility management. Findings indicated that the improved cooking stoves contributed to the restoration of trees on the landscape and improved community livelihoods (FAO, 2017b).

Improved ecosystem services have been reported in connection with nearly all of the SLM groups in a review conducted by Schwilch *et al*. (2014). SLM practices and technologies are capable of increasing production. They should be used to address increasing demands for food, fodder, and water. According to Akhtar-Schuster *et al*. (2011), SLM investments are an economically sensible way to address land degradation. Lovell and Sullivan (2006) indicated that vegetative filter strips reduced sediment yield, nutrients, and pesticides in surface runoff. In a study conducted by Mwangi *et al*. (2015) in Sasumua Watershed, Kenya, contour farming reduced surface runoff and sediment yield by 12 per cent and 24 per cent, respectively. SLM practices and technologies from the different groups work well independently but are even more effective when combined. A combination of contour farming implemented on the sloped areas and a grassed waterway implemented on the flat areas showed a combined reduction of sediment inflow to the reservoir by 66 per cent (Mwangi *et al*., 2015). Terracing combined with 10 m vegetative filter strips and grassed waterways reduced sediment inflow by 75 per cent (Mwangi *et al*., 2015). Terraces enhance infiltration and reduce surface runoff velocity, hence the water's erosive power (Arabi *et al*., 2008).

Therefore, implementing SLM practices and technologies in the Olifants sub-basin should restore degraded land, improve agricultural management, crops, livestock production, and increase water availability. The latter is essential to this study, aiming to assess the hydrological impacts of proposed SLM practices and technologies on the Olifants Sub-basin's water resources. In this sub-basin, SLM practices will be tested and possibly

adopted as part of a CSIR-led Global Environmental Facility (GEF5) SLM project to address land degradation due to improper land management and population pressure.

The SWAT model has been used around the world to assess the impacts of SLM or best management practices on the watershed hydrological status (Arabi *et al*., 2008; Fleskens, 2012; Schmidt and Zemadim 2013; Memarian *et al*., 2014; Mwangi *et al*., 2015; Eekhout and de Vente, 2019). Water resources managers have various management options. The effectiveness of each option in mitigating the negative impacts of land degradation on water resources needs to be assessed before implementation. That can be achieved with the aid of hydrological models. The evaluation of the impacts of implementing SLM measures is a cost-benefit analysis. It provides decision-makers with information on the effectiveness of specific measures before implementation, potentially leading to higher success rates (Mwangi *et al*., 2015).

1.2.Problem Statement

Land degradation is a long-standing threat to the environment, food security, and livelihoods. The Olifants Sub-basin is one of the most degraded water management areas in South Africa. As the result, it was selected as one of the sites within which innovative pilot SLM approaches will be implemented to address land degradation under the GEF5-SLM project. Sustainable land management (SLM) is a prominent tool in arresting land degradation issues. However, the adoption of SLM practices and technologies in the Sub-basin is significantly low. Low adoption rates result from the lack of an enabling environment that includes relevant policies and institutional capacity for significant adoption. The absence of reliable and regulated impact monitoring systems/studies and incentive measures to encourage SLM adoption in different land-use activities also affects adoption rates and upscaling. These constraints inadvertently promote inappropriate land-use practices that affect agricultural productivity, ecosystem services, and livelihoods. Therefore, there is a need for studies that assess and quantify the impacts of SLM practices and technologies before adoption. Such studies inform policymakers and catchment managers of valuable and useful SLM practices and inherently encourage land users to adopt and implement those practices. As a result, this study bridges this identified gap and provide incentives or results that would encourage agricultural communities, policymakers, water resources managers and the government to adopt and implement SLM practices and technologies.

1.3.Research Aims and Objectives

1.3.1. AIM

The study aims to assess the impacts of Sustainable Land Management practices and technologies on streamflow and sediment yield of the Olifants Sub-basin.

1.3.2. SPECIFIC OBJECTIVES

This aim will be achieved by addressing the following specific objectives:

a. To assess the impacts of proposed or identified SLM practices and technologies on water.
b. To assess the impacts of proposed or identified SLM practices and technologies on sediment yield.

1.3.3. RESEARCH QUESTIONS

This study will answer the following questions:

a. How can SLM practices and technologies be used to arrest land degradation in a semi-arid environment such as the Olifants sub-basin?
b. How does SLM practices and technologies affect water and sediment yield in the catchment?

1.4. Delineations – Limitations and Scope

The current study is an extension of the GEF5-SLM project funded by the United Nations Development Programme (UNDP) and the Council of Scientific and Industrial Research (CSIR). The Olifants Sub-basin is one of the three identified sites within which innovative pilot approaches to address land degradation will be implemented. The project team conducted stakeholder engagement workshops and field visits to get exposure to the villages severely affected by land degradation, engage with the local communities and key stakeholders and select appropriate SLM practices and technologies. The current study solely focuses on the impacts of selected or proposed SLM practices and technologies on hydrology. This study only looks at sediment yield and water yield as part of the hydrology. The study focuses on the SLM practices and technologies that were selected based on literature, field visits and stakeholder engagements. The research design and methodology focus only on quantitative methods to achieve the aim and objectives of the study.

1.5. The organisation of the thesis

The thesis is organised as follows. Chapter two will review the literature relating to the impacts of Sustainable Land Management (SLM) using examples from both developing and developed countries. The chapter also includes an overview of the Soil and Water Assessment Tool (SWAT) and its application. Chapter three describes the study area, the collection and collation of required data, and the processes involved in modelling the Olifants sub-basin with SWAT. Chapter four illustrates, interpret, and discuss the findings of the study. Chapter five concludes the dissertation, describes the limitations as well as recommendations.

2. LITERATURE REVIEW

The literature review chapter focuses on Sustainable Land Management (SLM), its principles and theory, SLM practices, technologies, and SLM impacts in different facets. Furthermore, the chapter will highlight the challenges and limitations of upscaling SLM and the different methods used to assess, quantify, or evaluate SLM impacts.

2.1 Definition and principles of Sustainable Land Management

Sustainable land management (SLM) is an umbrella concept that describes a range of techniques and practices that are related to land resources for socio-economic gains without compromising the environment's integrity. The origin of SLM dates back to the early nineties after the global discussion on sustainable development initiated by the Brundtland Commission. However, traditional land management practices related to SLM have always been around. The SLM concept was established at the 1992 Earth Summit and was first used in a published document by Smyth and Dumanski (1993).

SLM has been defined in different contexts, case studies, and by different authors and organisations using words to fit their respective fields. Hurni (1997) defined SLM as a system of technologies and planning that aims to incorporate ecological, socio-economic, and political principles to manage land for agriculture and ensure equity for future generations. The FAO defines SLM as a land-use system that ensures continuous environmental functions and productivity of resources that provide food to meet changing human needs. SLM is a way of increasing socio-economic benefits from land resources without compromising its ecological support functions through sustainable practices (TerrAfrica, 2005). In the African context, SLM is understood as a knowledge-based management system. A system that combines land, water, biodiversity, and environmental management to meet the ever-increasing food and fibre demands without compromising ecosystem services and livelihoods (Archer *et al*., 2018).

Motavalli *et al*. (2013) discussed the essential components of SLM, which include:

- understanding the ecology of land management;
- maintenance and enhancement of land productivity;
- maintenance of soil quality;
- increased diversity for higher stability and resilience;
- provision of economic and ecosystem services benefits for communities, and;
- social acceptability.

According to Motavalli et al. (2013) for any land use practice to be considered sustainable it has to embody the objectives of SLM. The objectives suggest that land management practices must ensure productivity, long term security, environmental protection, viability (must be effective) and must be socially acceptable.

Definitions of SLM vary according to the different perspectives of different contexts, stakeholders, or organisations. They are often influenced by the perception of scale and contextual factors. However, SLM's definition in different contexts and time scales has remained centred on sustainability principles, such as future generation equity and the balance between socio-economic progress and environmental protection. Even though different authors have used different wording and concepts, three essential principles of SLM remain constant. Those principles are known as integrative land management, environmental protection, and addressing socio-economic demands. SLM is a land-use system (TerrAfrica), a knowledge-based system (World Bank), or a practical tool (United Nations). SLM is all of that and more; it can be context-specific and remodelled to address local environmental issues. It is essential to foster SLM practices and technologies that cater to local issues based on a holistic and participatory approach that includes all stakeholders within an enabling environment (Hurni, 2000).

2.2 Sustainable Land Management.

Sustainable land management technologies are the practices used to ensure environmental protection, control land degradation, and enhance productivity in the field. The technologies can either be agronomic, vegetative, structural, or management measures such as land-use change, area closure, and rotational grazing (Schwilch *et al.*, 2011). The technology can be a combination of measures. Agronomic technologies include intercropping, contour cultivation, and mulching, whereas vegetative technologies consist of tree planting, hedge barriers, and grass strips. Structural technologies include measures such as graded banks or bunds and level bench terraces.

SLM practices should be flexible and multifunctional in approach to play a significant role in mitigating land degradation and restoring natural resources with significant onsite and offsite impacts (Teka *et al.*, 2013; McDonagh *et al.*, 2014). Many factors influence or incentivise farmers or land users, in general, to adopt and implement SLM practices. Farmers are likely to adopt SLM practices based on their farm household's benefits and abilities and farm quality and attributes (Kassie *et al.*, 2010).

Liniger *et al.* (2011) group the SLM technologies into eleven specific groups according to the familiar and standardised WOCAT format for documenting and disseminating SLM. The groups include rainforest, forest in drylands, planted forest, pastoralism and rangeland, integrated crop-livestock, agroforestry, cross-slope barriers, smallholder irrigation, rainwater harvesting, conservation agriculture, and integrated soil fertility management. However, for this study, SLM practices and technologies are grouped into five categories: water management, forest management, slope–barriers, rangeland management, and crop management. Such grouping is simple and more generalised because closely related technologies or practices share the same group. Thus, sustainable rainforest management, sustainable forest management in drylands, sustainable planted forest management form a broader group, namely forest management. Pastoralism and rangeland management and integrated crop-livestock management are collectively known as rangeland management. The crop management group encompasses agroforestry, conservation, and integrated soil management. Smallholder irrigation management and rainwater harvesting comprise the sustainable water management group.

2.2.1 WATER MANAGEMENT

Water management means efficient use and protection of all water resources, including surface, ground, and rainwater. It is paramount to integrate water management practices and technologies with improved soil, nutrient, and crop management to enhance land productivity and yields in small-scale agriculture (Rockström *et al.*, 2009). There are many practices and technologies in water management that can be reviewed. However, for this study, only four water harvesting techniques and smallholder irrigation management will be discussed. Water management practices reduce environmental risks and improve crop yield while simultaneously delivering positive impacts on other ecosystem services. Indirectly the practices increase local biodiversity, improve soil conditions, and promote socio-economic benefits.

Water Harvesting (WH) is the collection, concentration, and management of rainwater or floodwater runoff to make it available for domestic and agricultural use and ecosystem sustenance (Mekdaschi-Studer and Liniger, 2013). It is a relatively low-cost, simple technology with a proven capacity to improve the productivity of semi-arid zones (Bouma *et al.*, 2015). As countries face water scarcity, WH systems are receiving attention again after being abandoned and forgotten. The practice of water harvesting has been around and used by people, particularly those living in drylands (Marques et al., 2016). WH practices include floodwater harvesting, micro-catchment, micro-catchment, and

rainwater harvesting. Floodwater harvesting is a form of water management where floodwater or runoff from intermittent streams is intercepted and diverted for agriculture and domestic use. Small earthen or stone-built bunds intercept floodwater from intermittent streams. This process slows water flow, increases infiltration into the Soil, and diverts a portion of the floodwater to adjacent fields for irrigation (Marques *et al.*, 2016). Therefore, water that would be lost to evaporation and runoff without any benefit is kept.

Macro-catchment water harvesting (*Figure 2.1*) involves using small earthen dams to store the upstream water used for irrigation later (Marques *et al.*, 2016). The size of earthen dams can vary between 50 to 100 m long by 4 to 8 m deep in Zambia (Malesu, 2013). Small earthen dams provide water for domestic use, increase crop yields, and establish a water storage system for livestock, particularly in Africa (Marques *et al.*, 2016).

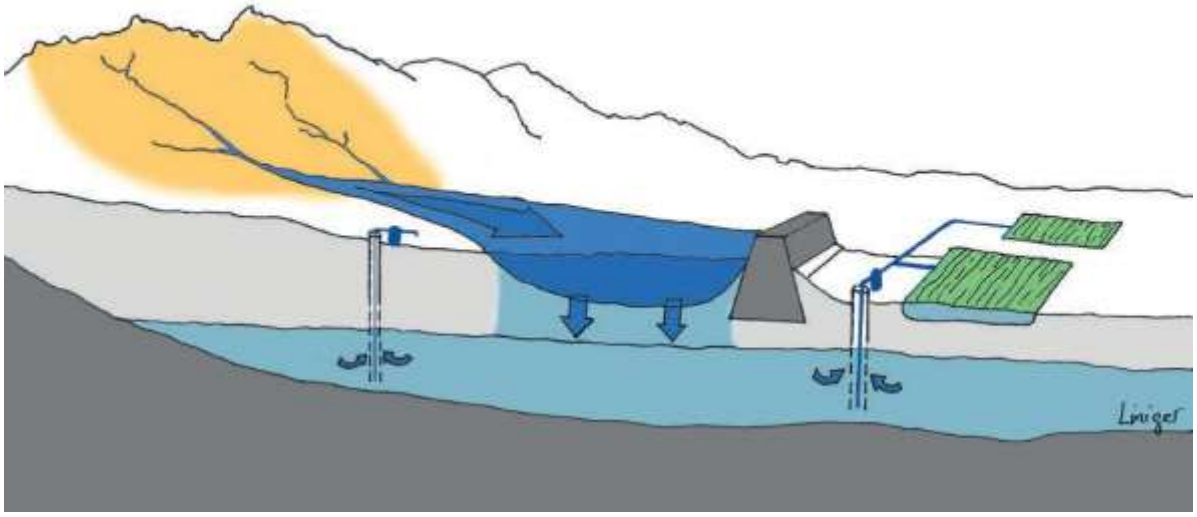


Figure 2.1. Macro-catchment rainwater harvesting system (https://www.ruvival.de/macro-catchment-rainwater-harvesting/)

Micro-catchment rainwater harvesting is a technique used to collect overland flow, increase water infiltration, and prevent soil erosion from small catchments of short lengths (Marques *et al.*, 2016). This type of water management includes practices such as planting pits and stone lines and furrow-enhanced runoff harvesting. Small pits are constructed during the dry season by digging out soil to increase rainfall infiltration for millet and sorghum crops and other crops. In Burkina Faso (*Zai*) and Niger (*tassa*), small pits are holes of 20-30 cm diameter and 20-25 cm depth, spaced about 1 m apart in degraded croplands (Zougmore *et al.*, 2003). This practice enables farmers to use small quantities of rainwater, manure, and compost very efficiently and rapidly restore degraded lands'

productivity (Hassane *et al.*, 2000). The techniques are cheap and time-efficient for addressing highly degraded areas (Ouedraogo and Sawadogo, 2001). Biazin *et al*. (2012) state that micro-catchment rainwater harvesting techniques are more common than rainwater irrigation techniques from macro-catchment systems. Depending on rainfall patterns and local soil characteristics, appropriate micro-catchment techniques could improve the rooting zone's soil water content by up to 30 per cent (Biazin *et al*., 2012).

Rooftop rainwater harvesting is a WH system designed to capture and store rainwater in closed storage facilities like tanks and drums. This method improves household access to water for drinking, sanitation, and home garden irrigation in dryland areas (Marques *et al*., 2016). Findings from other studies show that rainwater from rooftops is drinkable. However, water quality concerns are still related to chemical and microbial contaminants (Mwenge Kahinda *et al*., 2007). A study by Malema *et al*. (2018) indicated a prevalence of the pathotype neonatal meningitis *Escherichia coli* in the harvested rainwater. Such findings indicate that the collected rainwater is unfit for human consumption based on the set water quality guidelines. Water storage facilities should be covered, cleaned, and regularly treated with disinfectants such as chlorine for rainwater from rooftops to meet the international guidelines of drinking water quality (Marques *et al*., 2016). There is a need to raise awareness and implement sanitation methods to escalate this technology's uptake and dissemination.

Smallholder Irrigation Management (SIM) is another water management practice to address water scarcity issues, particularly on farm fields. SLM uses water efficiently to increase the number of crops watered by a single drop of water. SIM advocates the principle of 'more crop per drop of water. More efficient water use is achieved through more efficient water collection and abstraction, water storage, distribution, and water application to the field (Gurtner *et al*., 2011). A SIM unit refers to a plot of land (less than half a hectare) managed by an individual land user or by a group of farmers or community (Gurtner *et al*., 2011). Nakawuka *et al.* (2017) defined smallholder irrigation as an irrigation scheme managed by smallholder farmers (who own less than 2 ha of plots) or communities.

In the South African context, the smallholder irrigation scheme is defined as irrigation schemes built for and used by Black farmers (Van Averbeke *et al*., 2011). The latter definition has a different take, which includes historical race marginalisation. In contrast, other definitions are centred on the amount of land occupied and the management personnel. The inclusion of race in this definition is likely due to the former political system of South Africa, which discriminated against black people. Nakawuka *et al.* (2017) reported

that smallholder farmers abstract irrigation water from streams, rivers, lakes, groundwater, and ponds. Collected rainwater is the primary source of water in SIM. Other sources are only considered during dry spells and extended growing periods (Gurtner *et al*., 2011). Pumps, wells, gravity feed, plastic pipelines, rural aqueducts, and tanker trucks are used to abstract and convey water from these water sources.

2.2.2 FOREST MANAGEMENT

A forest provides habitats for animals and is a source of livelihood for humans. It contributes to food security, climate change mitigation and adaptation and protects the natural capital. When this natural resource is over-exploited and cleared for anthropogenic activities such as agriculture, it leads to biodiversity loss and reduction of its ecosystem services. The process is called degradation. The forest's capacity to provide essential goods and services to people and the environment diminishes. Afforestation and agroforestry, amongst other practices, can counteract degradation and improve forest ecosystems.

Afforestation is the establishment of a forest in an area where there was no previous tree cover. Afforestation includes soil preparation, weeding, planting, watering, fencing, and enclosure periods lasting a minimum of 6 years (Chakel, 2012). In Cape Verde, only 300 ha of land cover was forest land in 1975. After afforestation, over 90000 hectares of land was classified as forest (De pinas Tavares, 2012). Establishing a forest increases positive effects on the environment. It increases vegetation species and richness, water filtration, and groundwater level. Simultaneously, decreasing soil erosion, runoff, crusting, downstream flooding, siltation, and increasing soil structure, biomass, soil moisture, and soil organic matter. Afforestation activities should include local communities from the very beginning of the project to implementation. Such a bottom-up approach ensures that there is enough land left for agricultural activities.

Agroforestry is a land-use system that combines shrubs and trees with crops or animals on the same land unit. Together these components generate economic, environmental, and social benefits (Marques *et al*., 2016). The underlying principles and traditional knowledge of agroforestry have been forgotten and downtrodden around the world (Nair *et al*., 2017). A variety of agroforestry systems are used worldwide, including silvopastoral, agro-silvicultural, and agrosilvopastoral systems. Silvopastoral systems refer to the production of livestock and woody species (Baets *et al*., 2007). Agrisilvicultural systems combine woody plant species and seasonal plants (Baets *et al*., 2007). In contrast,

agrosilvopastoral systems involve livestock production, woody plant species, and seasonal plants in the same area.

Agroforestry also includes alley cropping, multilayer tree gardens, home gardens, and versatile trees on cropland. Shelterbelts, where widely spaced rows of trees are planted between annual crops or windbreaks and buffer strips in different ways, are also part of agroforestry. It helps diversify and sustain the broad spectrum of agricultural commodities for enhanced economic, environmental, and social benefits by integrating trees on farms and agricultural landscapes (ICRAF, 2008). An example of the complex impacts of agroforestry is the home gardens on the sub-montane slopes of Mount Kilimanjaro. The Chagga home gardens are home to various plant species, such as indigenous species from the former sub-montane forest for firewood, timber, and livestock fodder (Hemp, 2005). The indigenous tree provides shade for coffee trees cultivated as a cash crop. The traditional Chagga home gardens also consist of cash crops such as coffee, bananas, taro, or yams, and stall-fed animals in continuous and diversified production over the years (Hemp, 2005). Rehabilitation strategies such as planting new cash or food crops, considering the appropriate spacing of each species, manuring crops with dung from livestock, lopping fodder trees or shrubs, maintaining and improving irrigation furrows and terraces, and improving apiculture are some of the options being carried out in the home garden (Hemp, 2005). Long-term community involvement and improvement of advisory services are crucial to the success of such activities (Marques *et al*., 2016).

2.2.3 SLOPE-BARRIERS

Cross-slope barriers are structural and vegetative measures on steep lands. These can be in the form of earth or soil bunds, stone lines, barriers in gullies, and vegetative strips. They are best known for reducing slope steepness or length, which reduces runoff velocity and soil loss, thereby contributing to soil and water conservation (Schwilch *et al*., 2014; Marques *et al*., 2016). A vegetative filter strip is an area of herbaceous vegetation that traps sediments from overland flow (Mwangi *et al*., 2015). This practice can be implemented in various land-use systems but is primarily used to restore degraded croplands. Cross-slope barriers also include vegetated earth-banked and terraces. These prevent erosion and downstream flooding; thus, improving soil fertility and water supply.

Terraces are structural measures installed on sloping land to allow runoff water to infiltrate into the soil, thus reducing surface runoff and erosion. Well constructed and well-maintained terraces combined with nutrient application and the use of quality seeds are more likely to be implemented in the fields with moderate slopes (Wickama *et al.*, 2014).

SLM measures implemented solely and in isolation from other measures are less effective compared to combined SLM measures. *Atriplex hallmus* and aloe vera plant species are commonly used as living barriers to prevent soil erosion and improve soil conditions (Laouina, 2012). These can be combined with stone walls on the steep slope.

2.2.4 RANGELAND MANAGEMENT

Grazing land includes any area of pasture, rangeland, or grassland available for stock to graze. Rangelands are natural or near-natural ecosystems that comprise approximately 40 per cent of the earth's ice-free terrestrial surface (Sayre *et al.*, 2013; Wrobel and Redford, 2010). Natural grasslands, savannas, shrublands, deserts, tundra, alpine communities, and coastal marshes are rangelands. Rangelands can also include areas or lands that are revegetated naturally or artificially to provide plant cover that is managed like native vegetation (Marques *et al.*, 2016). These areas produce various ecosystems, such as regulating climate, livestock (meat) production, and carbon sequestration. However, Rangelands are degraded and under threat due to over-exploitation and overgrazing in the last several centuries (Walker, 2010).

Rangeland restoration techniques include passive and active strategies. The former include grazing enclosures and rangeland resting (Schwilch *et al.*, 2012; King and Hobbs, 2016). Practical approaches to land restoration include managed rotational grazing and improved *well* (water supply) distribution. These allow for water access, control and reduction of shrub encroachment, vegetation reseeding, and a succession of native species (Liniger et al., 2011; Kinyua *et al.*, 2010; Le Houerou, 2000). Papanastasis (2009) asserts that rangeland restoration's success relies first and foremost on improved grazing land management. Reed *et al.* (2015) argue that adequate policy support, coupled with economic mechanisms, could help reorient degraded rangelands towards more sustainable land management.

2.2.5 CROP MANAGEMENT

Soil and land degradation have led to a decline in the number of arable lands, which have significant food security threats. Practices that increase productivity on existing farmland and restore degraded arable lands are essential. Cropping management practices include those that increase soil fertility and crop production. These include organic and inorganic plant nutrients, conservation agriculture, and structural and vegetative measures. Practices that increase soil fertility include manure and compost or mineral fertilisers, increasing yield (Otinga *et al.*, 2013). These practices also have inherent benefits such as enhanced

soil structure, which increases microbial biomass and improves water filtration (Marques *et al*., 2016).

The use of green manure and low quantities of animal manure has been shown to increase soil fertility and biodiversity and reduce surface runoff and soil erosion (Marques *et al*., 2016). Products grown under the ecological agriculture system have a higher market price than products grown under conventional production schemes, which disincentivises land abandonment (Marques *et al*., 2016). Soil fertility can also be enhanced by producing fodder crops such as maize, oat, and wheat and rotating them. According to Marques *et al*. (2016), the dense surface cover of fodder crops protects the soil from wind and water erosion, increases soil fertility, improves plant and habitat diversity, and reduces soil salinity. These practices have the potential to improve soil fertility. However, Sutton *et al*. (2013) argue that agricultural soils in many developing countries are nutrient scarce, and many smallholder farmers lack access to affordable mineral fertilisers.

Minimum soil disturbance, crop rotation, and permanent cover enhance soil fertility and structure. These practices form a group known as Conservation Agriculture (CA). Some scholars note that the use of herbicides in CA can be controversial (Sanderson *et al*., 2013; Syswerda and Robertson, 2014). Despite these comments, the use of CA is growing across the world. CA reduces fuel and labour costs while reducing erosion, improving soil structure, and controlling weed (Marques *et al*., 2016). However, CA adoption in Africa remains relatively low compared to western countries, with the USA leading at 23 per cent of arable land under CA (Marques *et al*., 2016).

CA practices adopted in central Kenya, where a large scale commercial cereal farm is based, tractor-drawn equipment show increased yield after 20 years (Kisima, 2011). Marques *et al*. (2016) state that CA provides multiple socio-economic benefits and ecological onsite and offsite benefits related to soil, water, and biological conservation. Like other SLM practices, CA needs to be targeted and adapted to specific biophysical conditions, particularly conditions related to climate, topography, soil drainage, and soil texture (Marques *et al*., 2016). CA advocates minimum tillage, soil protection by vegetation residues and rotation and can also incorporate micro-harvesting techniques and micro fertilisation. Tmowlow et al. (2008) reported an increase in yield by 50 to 200 per cent, including improved soil and water quality in semi-arid areas of Zimbabwe as a result of CA practices.

2.3 Impacts of Sustainable Land Management

The impacts of sustainable land management (SLM) are multifaceted and all-around positive. Biophysical factors play a vital role in the effectiveness of certain SLM technologies and practices, which have not been addressed in the SLM adoption discussions (Letroy *et al.*,2000). They also influence the type of SLM measure that can be adopted and how effective it will be in addressing land degradation. For example, in the West Usambara highlands, soil conservation measures were recommended and adopted based on the steepness of the slope. Conservation measures such as terraces are usually implemented in fields with moderately steep slopes and are more effective in such areas. However, the adoption of these measures has been little because of other biophysical conditions such as soil type, field size, and cropping system, which play an essential role. Kassie *et al.* (2010) indicate that using minimum tillage in areas with low moisture and agricultural potential is more effective than high moisture and high agricultural potential areas.

Amid the mixed result from various SLM impact studies (Benin, 2000; Holden et al., 2001; Pender and Grebremedhin, 2006; Kassie and Holden, 2008), there is a need for careful and location-specific analysis of SLM practices and technologies. Biophysical factors such as rainfall and water availability influence SLM practices' effectiveness on land degradation and agricultural productivity (Kassie *et al.*, 2010; Wickama *et al.*, 2014). The impacts and effectiveness of different SLM practices and technologies largely depend on the type and magnitude of investments (Schmidt and Zemadim, 2013). The area-wide approach yields the most significant long-term benefits compared to individual farms; however, it is too expensive (Schmidt and Zemadim, 2013).

Most studies generalise the impacts of certain practices and technologies without being specific about their types. For example, studies investigating soil conservation impacts on agricultural productivity do not help to optimise the adoption and upscaling of SLM (Kassie *et al.*, 2010). Because such studies lack details about the exact type of soil conservation measures that enhances productivity. They also fail to detail the biophysical conditions or agro-ecological areas, which weakens SLM policies. Each SLM practice and technology have inherent benefits, trade-offs, and shortcomings concerning labour inputs and incurring costs (Schmidt and Zemadim, 2013). These should be stipulated to strengthen policy documents and ensure the application and scaling up of SLM practices and technologies.

2.3.1 LAND – SOIL AND AGRICULTURAL PRODUCTION

In this case, land refers to the terrestrial surface not covered by water and provides an agricultural production environment. Sustainable land management (SLM) aims to, amongst other things, improve the quality of land resources, which includes soil, water, and air. However, for this section, the focus is on soil and agricultural production. Fields with SLM measures have the deepest topsoil (>10 cm) due to good soil and water conservation practices that conserve soil moisture and increase soil structure. Improved soil structure and high soil water content stabilises soil particles and reduces erosion. Clay content increased in fields with proper SLM measures compared to fields with no SLM measures. A decreasing trend in organic content prevailed (Wickama *et al.*, 2014). Findings from Wickama *et al.* (2014) indicate a decrease in soil loss from fields with soil and water conservation measures. These findings indicate that using terraces and other physical structures (bunds, tree cover) and soil management practices (contour ploughing) reduce soil erosion. The results are in line with Tenge (2005) findings, which reported soil losses of 0.2-ton ha^{-1} y^{-1} from the terraced field in the Kwalei Catchment in Lushoto, Tanzania.

Soil erosion is at the centre of water quality management in many SLM studies. A lot has been done to underpin the principles and processes of sediment production and appropriate management practices to address the problem. However, sedimentation remains a significant and expensive water quality problem in many areas due to management initiatives that do not focus on the most critical sediments production areas. Studies have shown that contour farming has positive impacts in reducing erosion and water pollutants, particularly in agricultural lands (Arabi *et al.*, 2008, Stevens *et al.*, 2009, Mwangi *et al.*, 2015). Terraces are usually implemented in sloping land to make long slopes shorter. They reduce soil erosion by increasing infiltration and reducing surface runoff (Mwangi *et al.*, 2015). Terrace and bund construction in an area-wide approach plays a vital role in reducing surface runoff and erosion. As a result, sediment yield is limited (Schmidt and Zemadim, 2013).

Agricultural fields with SLM measures (i.e., well-constructed bench terraces and farmyard manure) had less erosion and higher crop yield (Wickama *et al.*, 2014). The yield of maize and beans was highly influenced by SLM measures with significantly high yields from fields with proper SLM measures (Wickama *et al.*, 2014). The yield of maize and bean crops increased with farmyard manure application and topsoil depth. The crops negatively correlated with slope and soil loss. Using farmyard manure and deep topsoil depth lead to increased crop yield. Overall results of the comparison study indicated that investments in

suitable SLM measures (i.e., farmyard manure and bench terraces) reduce erosion and increase crop yield (Wickama *et al*., (2014).

Moreover, minimum tillage compared with commercial fertiliser and farmers' traditional practices is more productive in the low-potential agricultural areas, increasing the net value of crop production by about $54 and $112 per hectare in the Tigray region, and $33 to $0 per hectare in the Amhara region. These findings exceed the average net value of crop production in the Tigray highlands. The adoption of SLM practices increases crop production. However, minimum tillage has little to no significant crop productivity impact in the Amhara region's high agricultural potential areas. SLM practices work differently in diverse agro-ecological systems with different biophysical factors.

The reported increased production in low agricultural potential areas with minimum tillage is likely due to this SLM technology (Kassie et al., 2010). However, the authors also highlighted that moisture conservation in high agricultural potential areas with fertile and wet soils leads to waterlogging, weeds, and pests. Increased percolation associated with SLM investments may extend the growing period, directly affecting farmer livelihoods (Schmidt and Zemadim, 2013). Hengsdijk *et al*. (2005) reported slightly increased productivity during drier periods associated with SLM investments.

Several empirical studies have emphasised the impacts of different land management practices on production, particularly in developing countries (Kassie *et al*., 2010). However, these studies solely focus on soil conservation measures as productivity-enhancing technologies without considering other SLM measures such as the ones under the water management group. The impacts of SLM practices are heterogeneous. Studies that assess and quantify such impacts need to account for econometric issues (Kassie *et al*., 2010).

2.3.2 WATER RESOURCES

Water resources include sources such as rivers, streams, lakes, reservoirs, springs and groundwater and the different hydrological process that underlines them. Studies that analyse the impacts of watershed management on water sources hardly consider the hydrological process in detail, particularly in developing countries (Nyssen et al., 2010). Some impact studies use hydrological models such as SWAT to assess the impacts of SLM practices and technologies on water resources, whether before or after implementation (Nyssen *et al*., 2010; Schmidt and Zemadim, 2013; Mwangi *et al*., 2015). Various measures such as stone bunds, check dams, and abandonment of post-harvest grazing in the May ZegZeg catchment in north Ethiopia led to higher infiltration rates and reduced runoff

volumes. These practices and technologies permitted farmers to plant crops in previously active gullies (Nyssen *et al*., 2010).

Schmidt and Zemadim (2013) also reported a decrease in surface runoff and erosion due to increased infiltration in the presence of terracing and residue management in the Mizewa watershed. Implementing residue management and terracing reduces the effects of extreme events on runoff response (Schmidt and Zemadim, 2013). An area-wide scale approach improves infiltration, decreases surface runoff and erosion, increases groundwater flow, and shallow aquifer recharge (Schmidt and Zemadim, 2013).

2.4 Methods used to assess Sustainable Land Management technologies and practices.

The methods are classified as quantitative and qualitative. Qualitative methods refer to tools such as focus groups and interviews used to explore and understand underlying reasons, opinions, and motivations. In contrast, quantitative methods use statistical and mathematical tools to quantify a problem by generating numerical values. Some scholars combine both methods to assess and quantify a problem, and this phenomenon is referred to as triangulation. It is the most recommended methodology because findings from one type of study can be checked against the other type's findings.

2.4.1 Methodological Approaches

Schwilch *et al.* (2011) reviewed the methodological approaches used to evaluate SLM's impacts at a local, regional, and global scale. These methods included the World Overview of Conservation Approaches and Technologies (WOCAT) framework, Land Degradation Assessment in Drylands (LADA) initiative, Global Environment Facility (GEF) knowledge from the land initiative and the DESIRE project The WOCAT method assesses SLM's biophysical and socio-economic impacts at local, regional, and global scales using two questionnaires. The questionnaires allow insight into the benefits, advantages and disadvantages, economic impacts, and acceptance of technology based on experts' scores and available supplementary scientific data (Schwilch *et al*., 2011). Although the WOCAT tool allows users to assess SLM implementation's cost and impact, the questionnaires are time-consuming and subjective.

DESIRE was a five year-year European Union project developed to combat desertification using new conservation strategies. This project involved 28 partner institutions and was

implemented in 18 different study sites. The DESIRE initiative used a participatory approach to assess the environmental and socio-economic impacts of SLM strategies. The methodology combines the WOCAT tools with extensive stakeholder engagements and decision-support workshops and field trials and monitoring (Schwilch *et al.*, 2011). Such an approach permits in-depth analysis of SLM measures' causes and effects on the environment and people at a local and regional level. The method can only be used by a team of specialists from disciplines related to land degradation, land management, land use, and soil and water conservation, and this serves as a limitation. Schwilch *et al.* (2011) assert that the DESIRE methodology's effectiveness would be more favourable when applied in an integrative manner for the long term.

The LADA project developed a standardised and scientifically-based methodology to assess SLM practices' impacts at different spatial scales in collaboration with local stakeholders and communities. The LADA methodology entails field observations and measurements, interviews, analysis of livelihoods, costs, benefits, and impacts (Schwilch *et al.*, 2011). The methodology has been piloted in South Africa, Cuba, Senegal, China, Argentina, and Tunisia as part of LADA countries. The LADA methodology includes identifying key stakeholders, collecting secondary information, focus group discussion, and community mapping and wealth ranking. A reconnaissance site visit and transect walk with land users is then conducted and coupled with biophysical assessment (soil, water, vegetation, onsite, and offsite/landscape) and livelihoods interviews with land users and other households (Bunning *et al.*, 2011). The LADA methodology can be integrated with the WOCAT and DESIRE to assess land degradation and SLM and their impacts on ecosystem services at a national/regional scale (Schwilch *et al.*, 2011).

The GEF methodology uses a set of global indicators to assess the impacts of SLM achieved through projects funded by the Land Degradation Focal Area of the GEF. The five global indicators were selected through an expert consultation process. These indicators include land use/land cover, land productivity of various land cover and land use systems, water resources availability, human well-being, and income distribution (UNU, 2011). Formulating the contextual indicators (i.e., precipitation, extreme natural events, extreme non-natural events, population density, and market prices) at a project level is vital. These contextual indicators ensure that when assessing the impacts of SLM, trends not related to a project's intervention can be accounted for and results can be compared against global indicators (UNU, 2011). Indicators can be adjusted to represent specific local needs and conditions (UNU, 2011).

The GEF KM: Land methodology is a hybrid SLM framework. It combines the Millennium Ecosystem Assessment (MEA) concepts with the Driver-Pressure-State-Impact-Response (DPSIR) model. The MEA concept focuses on ecosystem services and human well-being dynamics. In contrast, the DPSIR is more focused on the biophysical processes and drivers of change (UNU, 2011). The first phase of the methodology entails collecting data within a defined project impact using different tools and from several sources such as existing national data sources, project database developed through engagements with stakeholders as well as from community focus group discussions, interviews, surveys, and field measurements for biophysical data (UNU, 2011; Schwilch *et al.*, 2011). The second phase includes comparing observed changes with areas with no SLM. As for the third phase, contextual factors such as rainfall, extreme natural events, human conflict, market access are monitored and assessed to determine their impacts. The fourth step is to create an enabling environment for policy development, improve institutional capacity across all levels, and improve informed management about SKN implementation. Finally, interpret project-based data together with national or subnational stakeholders (Schwilch *et al.*, 2011). The methodology is simple and made up of widely-known and pre-existing frameworks, making it more acceptable to the United Nations Convention to Combat Desertification community (Schwilch *et al.*, 2011). However, this methodology is only limited to GEF funded SLM projects.

2.4.2 Qualitative Methods

An analysis of the benefits of SLM practices for climate change adaptation and socio-economic impacts in Imo State Nigeria was conducted using structured questionnaires and oral interviews (Chikaire *et al.*, 2012). The collected data was then analysed using percentages, frequency distribution tables, and mean scores (Chikaire *et al.*, 2012). This methodology allowed the researchers to determine the respondents' socioeconomic status, identify the SLM practices of respondents, and the benefits of SLM practices on climate change adaptation. The findings showed that SLM practices increased crop yield while decreasing pests and crop failure. While this study reports a positive correlation between the implementation of SLM practices and climate change adaptation, there is limited conclusive evidence to support a direct cause and effect relationship between the SLM practices and the actual impacts on climate change.

Kassie *et al.* (2010) systematically explored the productivity gains associated with adopting minimum tillage and commercial fertilisers using household-and plot-level surveys. The surveys were conducted in the Tigray and Amhara highlands regions of Ethiopia. The

author analysed the data for robustness using non-parametric and parametric methods (i.e., propensity score matching process) to check results robustness. The propensity score estimation was used to balance the observed distribution of covariates across the plots of adopters and non-adopters (Kassie *et al*., 2010). The findings showed that minimum tillage increased crop productivity in the low agricultural potential areas compared to commercial fertilisers. However, in high agricultural potential areas, commercial fertilisers are superior to minimum tillage and farmers' traditional practices. Although experiments and plot scale assessment provide an excellent basis for testing SLM practices' benefits, the testing conditions are limited. The results are not significant enough to inform

2.4.3 Quantitative Methods

Wickama *et al*. (2014) carried out an experimental study to determine the biophysical conditions of fields with SLM measures and assess their impacts on land degradation. The study was conducted in the West Usambara Highlands of Lushoto District, north-eastern Tanzania, using 60 fields with different SLM categories. The fields were classified into five SLM categories, ranging from fields with no SLM measures to well-maintained fields with prime land and crop management measures. Biophysical variables in those fields were measured for 21 months; these included soil properties, climate data, soil erosion, and crop yield. The collected data were analysed for the significant difference using Two-way ANOVA, one-way ANOVA, and t-tests. The study clarifies the impacts of SLM measures on erosion and crop yield on a small scale. However, the number of fields and the period (21 months) taken to measure the biophysical variables is insufficient. Due to extreme climate variability in the long term. Studies conducted at a field scale significantly contributes to the understanding of the impacts of SLM measures. However, they are limited and insufficient for upscaling of SLM practices due to spatial heterogeneity at a large scale.

Environmental models are increasingly used to assess and simulate the impacts of sustainable land management at a large scale (Arabi et al., 2008; Fleskens, 2012; Schmidt and Zemadim (2013); Memarian et al., 2014; Mwangi et al., 2015; Eekhout and de Vente, 2019). Such models consider different biophysical properties and provide greater insight into how different practices and technologies improve water quality and quantity. More scholars are using models to assess SLM practices' impacts and effectiveness under the current and future climate conditions (Azari *et al*., 2017; Eekhout and de Vente, 2019). Azari *et al*. (2017) assessed conservation practices' effectiveness under changed climatic

conditions using the SWAT model to simulate climate change scenarios. The model results were good enough based on the calibration and validation coefficient of determination (R^2) and Nash-Sutcliffe efficiency (NS) values for flow and sediment. Eekhout and de Vente (2019) evaluated SLM's effectiveness in a large Mediterranean catchment under the projected moderate and extreme climate conditions (water scarcity) using the SPHY_MMF model. SPHY-MMF is a spatially distributed hydrological model coupled with a soil erosion model and incorporates the Normalized Difference Vegetation Index (Eekhout *et al.*, 2018). The author calibrated both the hydrological and soil erosion models using daily discharge and literature data for hillslope erosion data. However, the author did not present the calibration and validation results.

Most physical-based models are not holistic in the sense that they do not include socioeconomic factors as inputs. Nor can such models be used to assess the benefits of SLM on socio-economic factors. Such limitation calls for physical-based models coupled with models that represent the study area's socio-economic complexities. These models should include socioeconomic factors such as the cost of technology and distance to the market (Fleskens, 2012). The PESERA-DESMICE model integrates the process-based erosion prediction model (PESERA) with the spatially explicit economic evaluation model (DESMICE). The hybrid model assesses the impact of SLM measures on land degradation while evaluating their financial variability. Fleskens (2012) highlighted three bottlenecks related to the model assessment of SLM technologies, which includes (i) the lack of knowledge about the unique variations SLM investments costs; (ii) the time it takes to realise the biophysical impacts is not explicit and (iii) fails to include characteristics of individual land users and the scale at which the SLM technologies are implemented.

Although the limitations of physical-based models have been established, the Soil and Water Assessment Tool (SWAT) is of particular interest in this study. The SWAT model was used in this study to quantify the impacts of sustainable land management (SLM) practices on water and sediment yield. The SWAT model has been used extensively to evaluate the impacts of different sustainable land management practices and technologies. Schmidt and Zemadim (2013) assessed the effectiveness of terraces, bunds, and residue management in the Mizewa watershed of the Blue Nile Basin. The model calibration and validation results were satisfactory, particularly when simulated monthly compared to daily and weekly timescales. The results indicate that a landscape-wide application of terraces and bund construction will improve infiltration, decrease surface runoff, and decrease erosion in the Mizewa watershed.

Applying the SWAT model to African conditions is usually attended with challenges due to the lack of reliable and high-quality data for biophysical variables such as soil properties and sediment load. Mwangi *et al*. (2015) evaluated agricultural conservation practices' impacts on water and sediment yield using the SWAT model in a data-poor watershed. The model performance of this study was satisfactory. Results gave monthly streamflow coefficient of determination (R^2) values of 0.80 and 0.85, Nash-Sutcliffe efficiency (NS) values of 0.74 and 0.81, and PBIAS values of ±5 per cent ±6 per cent during the calibration and validation. The study signified the importance of assessing the effectiveness of SLM options in addressing their negative impacts on water quality and quantity before implementation. The approach helps policymakers to make informed and sound decisions. The SWAT model plays a vital role in evaluating the effectiveness of Best Management Practices before implementation. This approach saves time and money and provides an even better alternative to the trial-and-error approach, which carries many inherent risks. Bracmort *et al*. (2006) used the SWAT model to assess the impact of structural best management practices in different conditions on water quality in Black Creek Watershed in Indiana, United States.

Several SLM practices have been studied for their impacts on water and sediment yield using the SWAT model. These include vegetative filter strips, contour farming, grassed waterways, parallel terraces, bunds, and residue management (Schmidt and Zemadim (2013); Mwangi *et al*., 2015). These practices are simulated in the model by adjusting the relevant parameters in the SWAT model. Most SLM measures and technologies are presented by relevant parameters such as the SCS curve number (CN), Universal Soil Equation practice factor (USLE_P), Universal Soil Equation cover factor (USLE_C), Manning's roughness coefficient for overland flow (OV_N), to name a few. Waidler *et al*. (2009) published a guide that details the parameters for specific conservation practices in a SWAT model. A method that can represent conservation practices was developed and evaluated by Arabi *et al*. (2008). The accuracy of modelled impacts of SLM investments may be compromised by the watershed subdivision routines used to parameterise the SWAT watershed (Arabi *et al*., 2008).

2.5 Adoption and upscaling of Sustainable Land Management

The rate at which Sustainable Land Management (SLM) technologies are being adopted remains low, particularly in developing countries (Mwangi and Kariuki, 2015). Adoption means different things to different scholars and organisations. According to Loevinsohn *et al*. (2013), adoption integrates new technology into existing practice, usually preceded by

a period of 'trying' and some degree of adaptation. Adoption in this thesis refers to the adaptation, use, and implementation of SLM technologies and practices.

A significant number of SLM technologies and practices exist and have been outlined and assessed in many scientific papers on land restoration and rehabilitation in recent decades (Quillérou and Thomas, 2012). However, established scientific evidence of these technologies and practices' benefits is not enough to encourage adoption, mainly due to the lack of connection between science and practice. The lack of policies that advocate and recognise that landscapes are social-ecological systems also hinders the adoption of SLM practices (Gobster, 2014). Subsequently, the developed and established policies may be ineffective and inefficient, and policymakers may not be well-informed about the environmental and socioeconomic benefits of sustainability and habitat restoration (Fleskens *et al*., 2014). Schwilch *et al*. (2014) assert that policymakers must integrate the socio-economic, cultural, and environmental benefits of SLM with its trade-offs and off-site impacts.

There are so many factors that influence the effectiveness and adoption of SLM technologies and practices. According to Wickama *et al*. (2014), the slope is used to determine the types of SLM measures adopted in the West Usambara Highlands, Tanzania. Such an approach alludes to the findings of Ketema and Bauer (2012). They concluded that slope and farm size influence the cost of constructing soil conservation measures such as terraces. Slope steepness should not be used as a principal determinant for the adoption of SLM practices. Other biophysical conditions such as soil type, field size, and cropping system also play an essential role. According to Mascarenhas (2000) and Tengeet *et al*. (2007), a farmer's choice to adopt SLM measures is essentially a socio-economic decision. Isgin *et al*. (2008) assert that farm technologies' adoption depended on farm size, farmer demographics, soil quality, urban influences, farmers' status of indebtedness, and location.

According to Mwangi and Kariuki (2015), factors that influence or determine the adoption of SLM practices and technologies are technological, economic, institutional, and human-specific — referring to factors such as the characteristics of the technology, farm size, cost of construction, off-farm income and socio-economic status of a land user, i.e., education, age, gender, and household size. These factors can have positive and negative impacts (or both) on the adoption of SLM practices and technologies. Farmers who are active participants in community-based organisations are more likely to engage in social learning about SLM technologies and practices. Such individuals are more likely to adopt technologies (Katungi and Akankwasa, 2010). Slope and farm size were essential factors

that farmers considered before constructing soil conservation measures such as terraces in their field (Ketema and Bauer, 2012). Perhaps, it would be better if each SLM practice or technology incorporates all factors or conditions that need to be considered before adoption.

Lee (2005) asserts that difficulties characterise the adoption of SLM technologies due to their location-specificity. Most SLM practices strive or perform best in certain biophysical conditions; thus, generalisation and upward-scaling are limited. There is a lack of information regarding local biophysical factors' impacts on certain SLM practices' effectiveness in SLM adoption discussion. Limitation like these hinders the upscaling process since there is not enough knowledge to account for biophysical factors' impacts on SLM practices during implementation and evaluation. Biophysical factors are essential for both the performance and the effectiveness of SLM measures. However, it is unknown which specific biophysical factors dominate such SLM measures (Wickama *et al.*, 2014). Therefore, studying the effectiveness of SLM practices and technologies across different biophysical conditions is essential. It will add to knowledge and help promote effective practices and technologies that successfully reduce soil degradation in areas with similar biophysical conditions (Wickama *et al.*, 2014). SLM practices are usually adopted based on local traditional practices and incremental experimentations than scientific evidence (Stringer *et al.*, 2013).

Many factors contribute to SLM practices and technologies' limited upscaling (Kassie *et al.*, 2010). Scaling up means expanding, replicating, adapting, and sustaining successful SLM measures in geographic space and over time to reach many people. Flawed extension services; blanket promotion of technology to very diverse environments; top-down approach to technology promotion; late delivery of inputs; low return on investments; lack of access to seasonal credit; production and consumption risk affect upscaling (Kassie *et al.*, 2010). Although these factors are based on a specific case study, the same applies to other areas. Agricultural advisors' services include demonstrating the use of commercial fertiliser and improved seeds. However, this is not the case for SLM practices and technologies. Extension officers' training does not include the implementation of SLM practices in agricultural fields. As a result, practices are being promoted without educating farmers and land users about the technical skills needed to implement SLM measures. The promoted SLM measures also fail to account for agroecosystem variability. SLM measures are promoted as a blanket one-size-fits-all.

Spielman *et al.* (2010) state that the standardise package approach does not allow farmers to explore, experiment, learn, and adapt technologies based on their own needs. The

standardised approach gives rise to inappropriate technologies to local conditions and unacceptable to farmers, which infringe on SLM principles (i.e., must be socially acceptable). According to Keeley and Scoones (2004), conservation interventions through the blanket application in Ethiopia are supported by simplistic, often unjustified claims which often negatively affect poor people's livelihoods. Assessment of SLM's effectiveness at a local level fails to provide a holistic overview of the spatial extent and effectiveness of such practices on a larger scale due to spatial variation, off-site effects, and cross-scale interactions (Schwilch *et al.*, 2011). Thus, the local assessment of SLM cannot be extrapolated or upscaled horizontally and vertically. The use of models to predict the impacts of certain SLM practices at a large scale could suffice.

Extrapolation of lessons learned from land restoration projects is complicated by the diversity of practices available to land users and the diversity of plot conditions (Giller *et al.*, 2009). These also include labour requirements, access to herbicides and nutrients, climatic conditions, and market constraints. Successful approaches are critical to the upscaling of technologies over a larger area and more land users (Schwilch *et al.*, 2011). An SLM approach is the way and means of support that help introduce, implement, adapt, and promote those technologies on the ground, such as farmer-fields schools (Schwilch *et al.*, 2011).

2.6 Summary

Sustainable Land Management (SLM) is considered the key to addressing ongoing and contemporary environmental issues such as land degradation. It has been studied using both qualitative and quantitative methods or a mixture of both methods. Most SLM measures and technologies are presented by relevant parameters in the SWAT model. Therefore, the model is of particular importance in the current study as it allows for the simulation of SLM practices and technologies by adjusting relevant parameters. Furthermore, it is essential to understand the impacts of management practices on water resources before implementation. As to increase the potential success rate and help water resources managers make informed decisions. The SWAT model has been successfully used to assess and quantify the impacts of SLM investments before implementation. SLM practices and technologies have proven to have significant strides in alleviating degraded lands, improving water quality and quantity and increasing food production. Nevertheless, SLM adoptions are significantly low, partly due to the SLM targets and indicators being mostly site- and nation-specific and lacking coherent and evidence-based policy framework and adequate policy action for SLM. Therefore, it is crucial to give land users a

glimpse of the future through physical-based models capable of assessing SLM practices and technologies' likely impacts before adoption.

3. RESEARCH DESIGN AND METHODOLOGY

The research design of the study is based on a quantitative method, namely modelling. A qualitative method is also essential in SLM for gaining insight and understanding of the land user's perception of SLM practices and technologies. However, in the scope of the current study, a qualitative research method was deemed insignificant. The current study is part of the GEF5-SLM project in the Olifants Sub-basin led by the CSIR. Stakeholder engagement workshops and focused group interviews were conducted to identify and facilitate the selection of appropriate SLM practices and technologies to be implemented at the Lepellane catchment and then scaled up to the entire Olifants sub-basin. Thus, the current study is a part of that project and focuses primarily on assessing the impacts of the proposed and selected SLM practices and technologies on hydrology before implementation to ensure success rate.

The study employed the Soil Water Assessment Tool (SWAT) model to simulate and predict the impacts of SLM practices and technologies on streamflow and sediment yield. SWAT is a basin-scale simulation tool that has been used around the world to address water questions. The SWAT model was developed to predict the effects of different management practices on water quantity and quality, sediment yield, and pollution loading in basins (Memarian *et al.*, 2014). Numerous studies have demonstrated the significant advantages of using SWAT to assess land management impacts on hydrology (Niraula *et al.*, 2015; Mwangi *et al.*, 2015; Gyamfi *et al.*, 2016b; Woldesenbet *et al.*, 2016). The model can be used in different interfaces; for this study, the ArcSWAT ArcGIS extension was used.

3.1 Description of the study area

The Olifants is one of the sub-basins of the Limpopo River Basin. The Olifants River rises in the Highveld region of South Africa between Breyten and Bethal. It flows north through several impoundments and is forced east by the Transvaal Drakensberg. It cut through at the Abel Erasmus Pass. Then flows across the Limpopo Province to join the Letaba River. The total catchment area of the river is about 54,740 km^2. It spans between latitudes 23.8˚S and 26.5˚S and longitudes 28.3˚E and 31.9˚E. The Olifants River is approximately 770 kilometres long with seven main tributaries, namely Elands, Ga-selati, and Wilge Rivers (located on the left bank), Blyde, Klaserie, Steelpoort, and Timbavati (*Figure 3.1*).

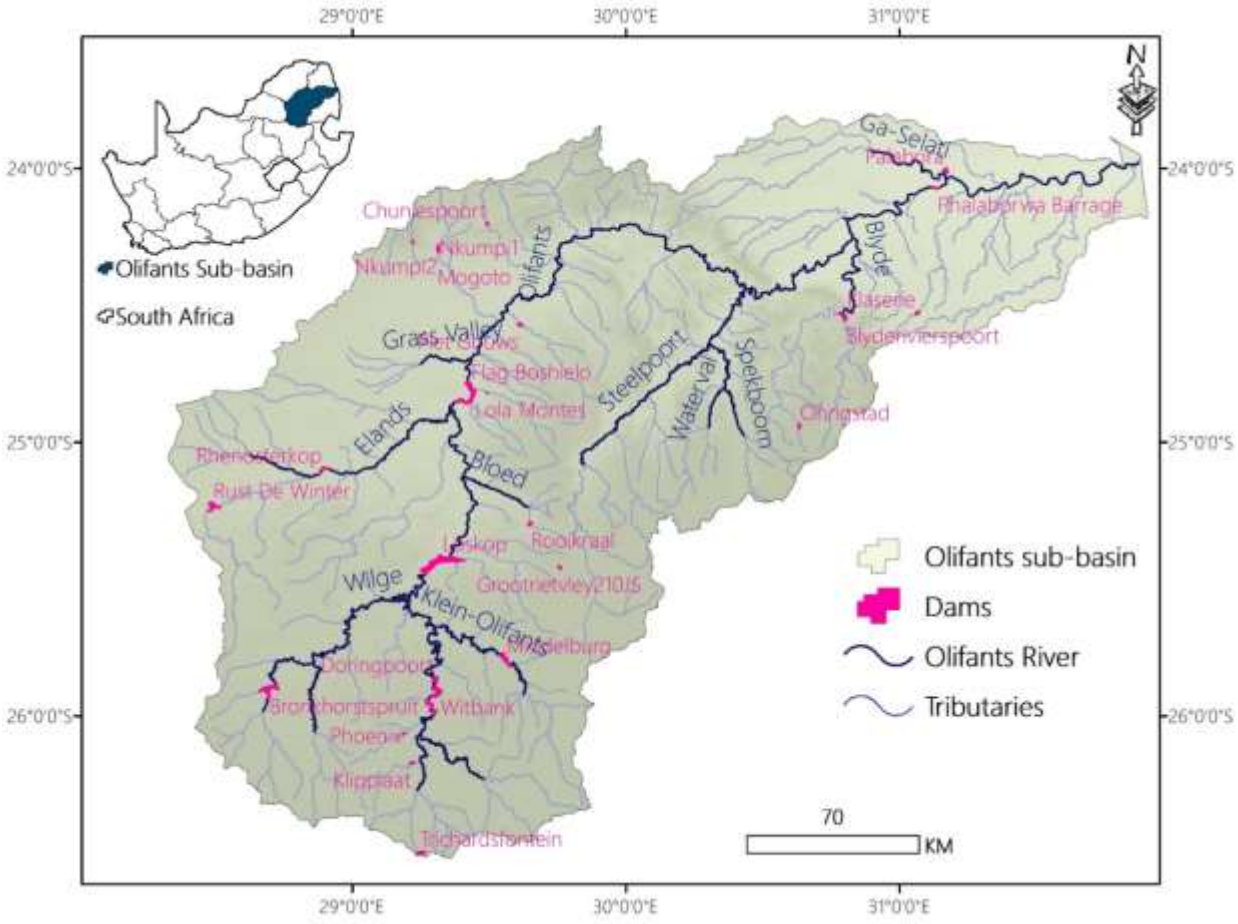


Figure 3.1. Map of the Olifants sub-basin and its location in South Africa.

According to the South African National Water Resources system, the sub-basin is divided into four hydrological sub-areas and 114 Quaternary Catchments (QC). QC refers to the central water management unit with a similar runoff (De Lange *et al.*, 2005). Eight catchments are of particular reference in this study since there are no dams or impoundments in their drainage area and have flow gauging stations with viable monthly average streamflow data. These include B11G, B12C, B20B, B42F, B60D, B71D, B72E and B73B. The classification of hydrological areas *(Figure 3.2*) is also associated with ecological regions such that:

- the Upper Olifants corresponds to the Highveld region,
- Upper Middle and Lower Middle Olifants is home to the Middleveld region,
- The Mountain area characterises the Steelpoort region,
- Moreover, the Lower Olifants represent the Lowveld ecoregion.

The Upper Olifants sub-basin (Highveld region) part of the Olifants has an average rainfall of 682 mm, higher than the Middleveld. Such climate conditions make the area more favourable to rain-fed cropping and stock farming, coal mining, and coal-fired power

generation (De Lange *et al.*, 2005). Undulating and aesthetic natural landscapes characterise the Middleveld region with water activities such as river rafting during high flows (De Lange *et al.*, 2005). The Middle Olifants is one of the poorest regions in the country. Land use activities in this region include irrigated agriculture with a variety of crops and large citrus plantations. The same occurs in the Mountain region. There are intensive irrigation and extensive mining activities in the lower riding areas. The Lowveld region is home to game farms and industrial activities closer to the Kruger National Park conservation area. The area is also characterised by low water quality due to effluent from nearby industries, a concern to conservationists in the area (De Lange *et al.*, 2005).

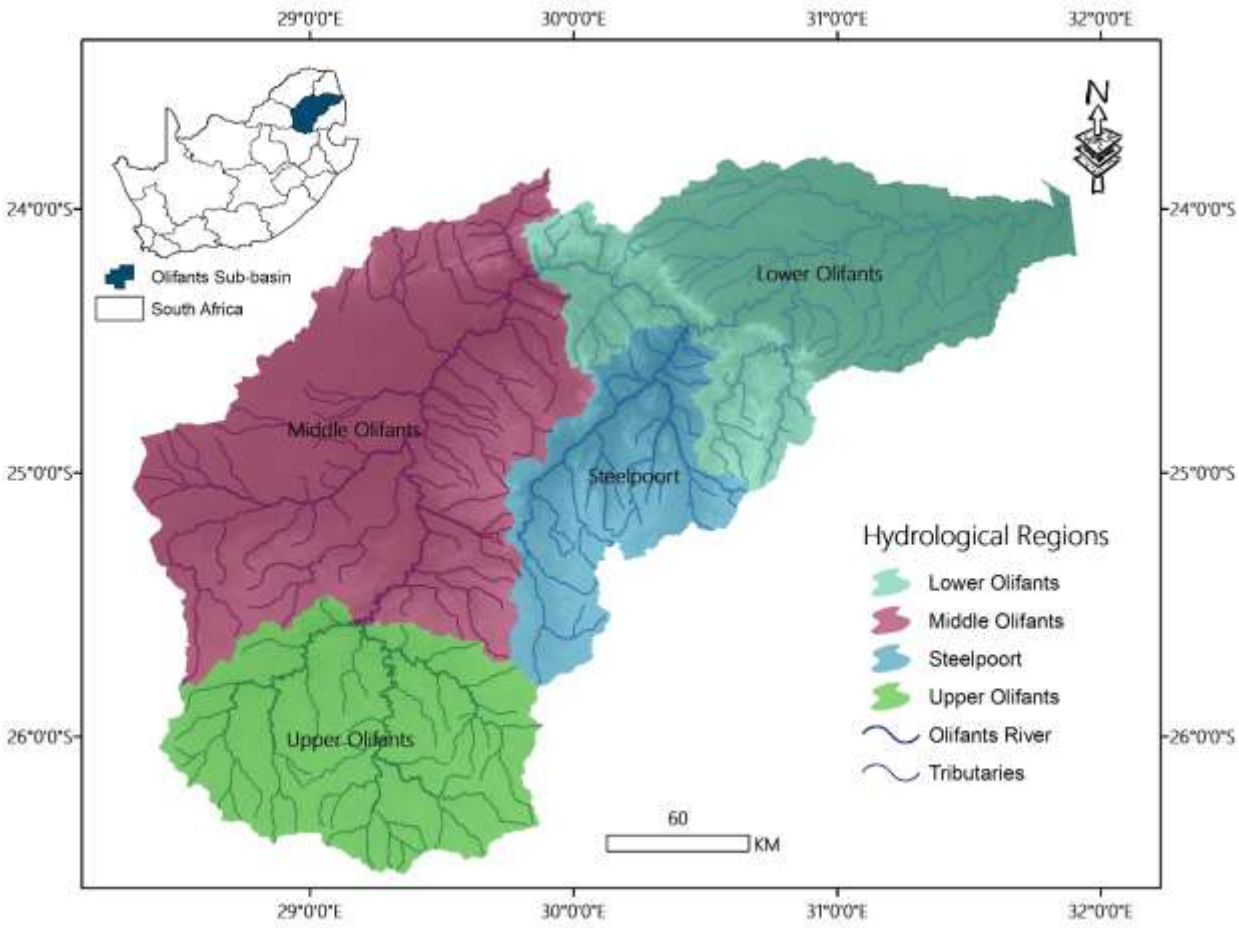


Figure 3.2. Main hydrological regions of the Olifants Sub-basin.

3.1.1 CLIMATE

The Olifants basin is located in a semi-arid climatic zone with annual precipitation of 630 mm, which peaks in January. The climate is semi-arid, with rain falling primarily during the summer (December to February). The sub-basin is characterised by the occurrence of orographic rainfall in the vicinity of the escarpment, which leads to the mean annual

precipitation exceeding 1000 mm in some areas (McCartney, 2003). The mountains run through the Sub-basin in an almost north-south trajectory; at the west side of the escarpment is Highveld (elevation > 1000 meters above sea level (masl)). The Lowveld is located on the east side with an elevation of less than 800 masl. The mean annual precipitation in both the east and west sides of the escarpment is approximately 600 mm (McCartney, 2003). There is a significant decline in annual total precipitation accompanied by an increase in rainfall intensity and heavy rainfall events. The potential evaporation in the Sub-basin is 1,700 mm. The mean rainfall only exceeds 50 per cent of mean potential evapotranspiration in 4 months of the year. As a result, the catchment is not suitable for the growth of crops. In many areas, irrigation is required to reduce the risks of water shortage.

The rainy summer season temperature ranges from 18ºC to 34ºC and from 5ºC to 26 ˚C in the dry winter season (Gyamfi *et al.*, 2016b). The Sub-basin temperature varies from -4 to 45 ˚C (De Lange *et al.*, 2005). There is a warming-up trend in the Sub-basin characterised by increasing daily minimum temperature and summer (days with greater than 25 ˚C) days in a year. Sing *et al.* (2013) alludes to an increasing and coherent trend in temperature in the Olifants Sub-basin. High temperatures lead to wat loss through evaporation, particularly from reservoirs.

3.1.2 TOPOGRAPHY

The basin's elevation ranges from 0-2400 masl, significantly influencing rainfall distribution and run-off in the Sub-basin. The slope gradient of the Olifants Sub-basin varies from flat (0-2 per cent) to very steep areas (>45 per cent). The dominant slope type is gently undulating slopes (*Figure 3.3*). The Mountainous hydrological areas of the Sub-basins have a steep slope (>16 per cent).

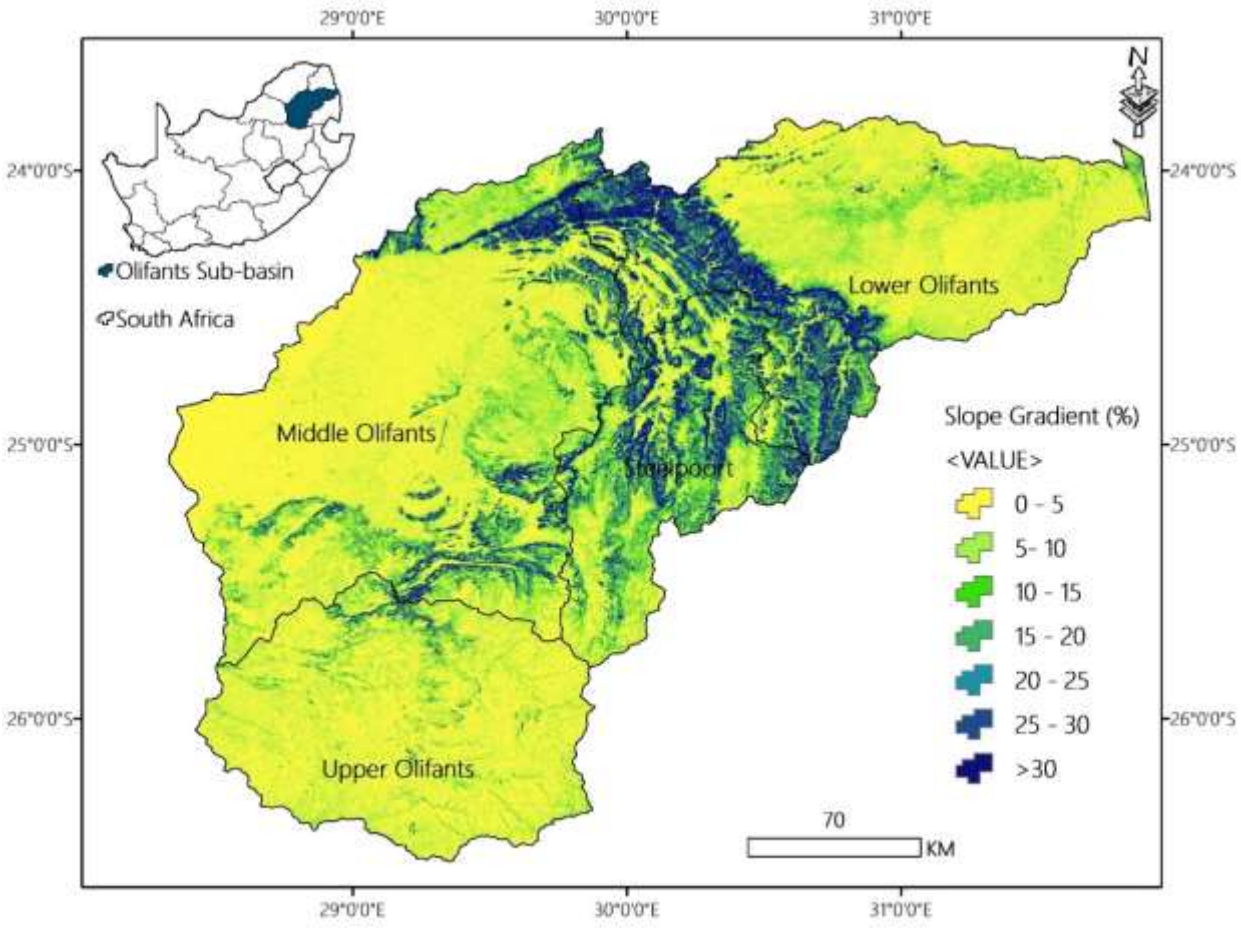


Figure 3.3. Slope gradient (in percentage) of the Olifants Sub-basin.

3.1.3 SOIL AND ERODIBILITY

According to FAO (2005), the most dominant soils types in the Olifants Sub-basin are cambic arenosols, chromic luvisols, chromic vertisols, orthic acrisols, rhodic ferralsols. Arenosols are sandy soils that originate from quartz-rich unconsolidated calcareous material or sandstone rock. Chromic luvisols have a high clay content in the lower horizons compared to upper horizons. Vertisols are heavy and internally moving clay soils with clay materials that swell when it comes to contact with water, resulting in large cracks when dry. Acrisols are developed from weathered acid and degrading rocks and have a higher clay content in the subsurface. These soils are dominantly found in the highveld of the Sub-basin. Rhodic Ferralsols are deeply weathered red or yellow soils that originated from basalts and diorites rocks. More information regarding the physical and hydrological properties of the dominant soil types, according to the FAO, is represented in Table *3.1* and Table *3.2*.

Table 3.1. The physical and hydrological properties of the dominant soils in the Olifants Sub-basin, according to FAO.

Soil Type	FAO Code	Hydrologic Group	Texture	Bulk Density (g/cm^3)	Available Water Content (mm/mm)
Cambic Arenosols	Qc24-1a	C	SaLm	1.45	0.08
Chromic Luvisols	Lc65-1-2a	C	SaLm	1.45	0.14
	Lc3-2ab	C	SaClLm	1.50	0.18
	Lc66-1a	C	SaLm	1.45	0.14
	Lc64-2b	C	SaClLm	1.50	0.16
Chromic Vertisols	Vc23-3a	D	Cl	1.60	0.14
	Vc1-3a	D	Cl	1.65	0.13
Orthic Acrisols	Ao69-1a	C	SaLm	1.5	0.01
Rhodic Ferralsols	Fr20-3bc	C	Cl	1.2	0.16

In the South African context, soil classification employs two methods that complement each other. These include the binomial classification by MacVicar *et al.* (1977) and taxonomic classification by the Soil Classification Working Group (SCWG) (1991). The former method identifies and names soils based on an orderly system of defined classes. Revealing the inter-relationship between soil properties and soils information is accurate and consistent (MacVicar *et al.*, 1977). The classification method indicates the nature of soils based on master horizons and diagnostic horizons. It provides a simplified soil series based on clay content and sand grade (Schulze *et al.*, 1995). The SCWG (1991) added categories of soils into families to represent the broader division of the soil series since the information needed at the series level of classification was not available.

Soils are made up of horizons formed under certain genetic conditions, which leads to some vertically stratified layers and easily identifiable and generalised master horizons (MacVicar *et al.*, 1977; SCWG, 1991). Master horizons' distinct properties were used to identify topsoil and subsoil diagnostic horizons based on the soil's pedological features. Water retention, free drainage, and conductivity were used to identify further topsoil diagnostic horizons. The identified diagnostic horizons were *organic*, *humic*, *vertic*, *melanic* and *orthic*. *The organic* horizon has high organic carbon content and the *humic* horizon is thick and rich in humified organic matter (MacVicar *et al.*, 1977). *Vertic* horizon has high clay content *whereas the melanic horizon is* dark coloured and well-structured whereas *the orthic* horizon is characterised by the absence of organic carbon, thickness, and expansive soils (MacVicar *et al.*, 1977).

Similarly, specific characteristics were used to indicate hydrological responses of diagnostic subsoil horizons such as (i) E horizon which is associated with periodic saturation and hence high interflow rates, (ii) G horizon is characterised by frequent

waterlogging, (iii) red apedal horizon which is structureless and well-drained, (iv) plinthic horizon this contains indurated iron pan zones of iron/Magnesium, indicating poor drainage and waterlogging potential, (v) prismacutanic horizon indicates abrupt increases in clay content and firmness associated with decreases in hydraulic conductivity or permeability, (vi) placic pan horizon represent an impediment to root development and water movement, (vii) dorbank horizon is characterised by a hard, massively structured, and silica cemented subsurface which is virtually impermeable and prone to waterlogging and (viii) hardpan carbonate horizon is a calcium/magnesium carbonate cemented hard layer that is slowly permeable to water and a barrier to root development (Schulze *et al*., 1995).

Soils were then grouped into 73 soil forms, each defined by a unique combination of vertical horizons and the grouping of certain kinds of diagnostic horizons in specific sequences. Soil forms were classified into soil families using 19 properties, which included thickness of the humic horizons, amongst other things. Soil forms were further subdivided into soil series based on soil texture, degree of leaching, calcareousness, surface crusting tendencies, cutan formation, and degree of wetness (Schulze *et al.*, 1995). To enable non-soil scientists (i.e., hydrologists) to describe soils according to criteria influencing soil's response and distribution of soil properties within their catchments, Land Types are the most comprehensive mapped soil information available in South Africa. A land type is a combination of similar terrain (topography), macroclimate, and soil pattern.

The Institute of Soil, Climate and Water (ISCW) of the Agricultural Research Council (ARC) embarked on a land type survey programme. The survey aimed to compile an inventory of natural factors that determine agricultural potential, define each profile with in-depth analysis, and delineate land types at 1:250 000 scales. A land type map and memoirs detailing soil information for each land type were produced (Appendix A). There are 332 land types within the Olifants. The land type memoirs were interpreted for hydrological modelling by Schulze *et al*. (1995). The scholars assert that soil properties are essential for hydrological interpretation and input into both water budgeting and physical-based models. These include depth of the top and subsoil horizons, soil texture, and soil water retention, amongst other things.

A method from Schulze *et al*. (1995) was adopted in this study to determine the depth of the top and subsoil. Depth in the Land type memoirs is given as a range per soil series. For this study, the most dominant soil series based on the areal percentage coverage was used. As a result, the depth range of that soil series was the total soil profile depth with a depth of 1 200 mm + assumed as 1 200 mm. Expert knowledge-based working rules of

Angus and Schulze (1990) were adopted in this study to partition the total depth into the top and subsoil horizons' depth. The method suggests that one add the lower and upper values of the depth range. If the sum is more than 600 mm, then the topsoil and subsoil constitute 0.33 and 0.67 per cent of the total depth, respectively. If the total depth is below 600 mm, the topsoil horizon is assigned a thickness of 200 mm and the subsoil the remainder.

According to the Land type map, the most dominant soil forms in the Olifants Sub-basin are Hutton (Hu), Shortland (Sd), Grenrosa (Gs), Mispah (Ms), and Arcadia (Ar). Hutton soil form has an Orthic A horizon overlying an Apedal (unstructured) Red B horizon. The soils are deep well differentiated. Shortland soil form also has an Orthic A horizon and a red structured B horizon. In contrast, Glenrosa soil form is characterised by an Orthic A horizon and a lithocutanic B horizon. It is mainly found in shallow rocky soils. Like Glenrosa soil form, Mispah soil form is shallow and has an Orthic A horizon overlying hard rock. Arcadia soil form. At a soil series level, dominant soil series include Argent (Sd11), Oribi (Gs), Hutton (Hu16), Msinga (Hu26), Mispah (Ms10), Eenzaam (Ar40), Glenrosa (Gs15), Robmore (Gs18), Bontberg (Hu25) and Clansthal (Hu25). *Table 3.2* presents more information regarding the soil properties such as bulk density, carbon content, and the hydrological group of the dominant soil series, adopted from Schulze (1995). Sandy clay loam soil texture is dominant in the Sub-basin, an excellent soil texture to grow crops. Hence, agriculture in the Sub-basin being one of the most dominant land-uses.

Table 3.2. The Physical and hydrological properties of the dominant soil series in the Olifants.

Soil Form	Soil Series	Texture	Hydrologic Group	Organic Carbon	Bulk Density (mg/cm^3)	Saturated Hydraulic conductivity (mmh^{-1})
Arcadia	Arcadia (Ar40)	Cl	C/D	0.38	1.29	0.6
Glenrosa	Glenrosa (Gs15)	SaLm	B	0.71	1.33	26
	Robmore (Gs18)	SaClLm	B/C	0.19	1.44	4.3
Hutton	Hutton (Hu16)	SaClLm	A	0.19	1.44	4.3
	Clansthal (Hu24)	SaLm	B	0.71	1.33	26
	Bontberg (Hu25)	SaLm	A	0.71	1.33	26
	Msinga (Hu26)	SaClLm	A	0.19	1.44	4.3
Mispah	Mispah (Ms10)	SaClLm	C	0.19	1.44	4.3
Shortland	Argent (Sd11)	SaCl	B	0.38	1.42	1.2

A report by Moolman *et al.* (1999) indicates that soil type, amongst other things, contributes to sediment production and erosion. Moreover, most of the Sub-basin dominant soil series have a moderate soil erodibility rate (Lorentz and Schulze, 1995). The

soil erodibility map in Moolman *et al*. (1999) indicates some areas with high soil erosion hazards. Particularly in the middle and lower hydrological regions of the Olifants Sub-basin. Moolman et al. (1999) research indicate that most non-point sediments originate from densely populated areas, steeper slopes, and more intense rainfall. Sedimentation in the Sub-basin is considered one of the primary environmental concerns; however, no preventative or monitoring measures are implemented.

3.1.4 Land Use/Land Cover

Land use types in the Olifants Sub-basin include agriculture, mining, forestry, animal husbandry, and settlements. The most dominant land use/land cover are agriculture, built-up areas, and vegetation in forests and rangeland (*Figure 3.4*). The land use/cover in the Sub-basin has undergone significant changes over time from converting rangeland and forest land to agricultural fields and residential areas (Gyamfi *et al*., 2016b). Agriculture and other economic activities in the Sub-basin, such as mining, contribute to the Gross Domestic Product of South Africa. There are extensive coal mining activities in the Upper Olifants for export to Richards Bay industries and use in the active coal-fired power stations in the area. The presence of coal mining in the area led to establishing the steel manufacturing industries located in Middelburg and Witbank (Upper Olifants).

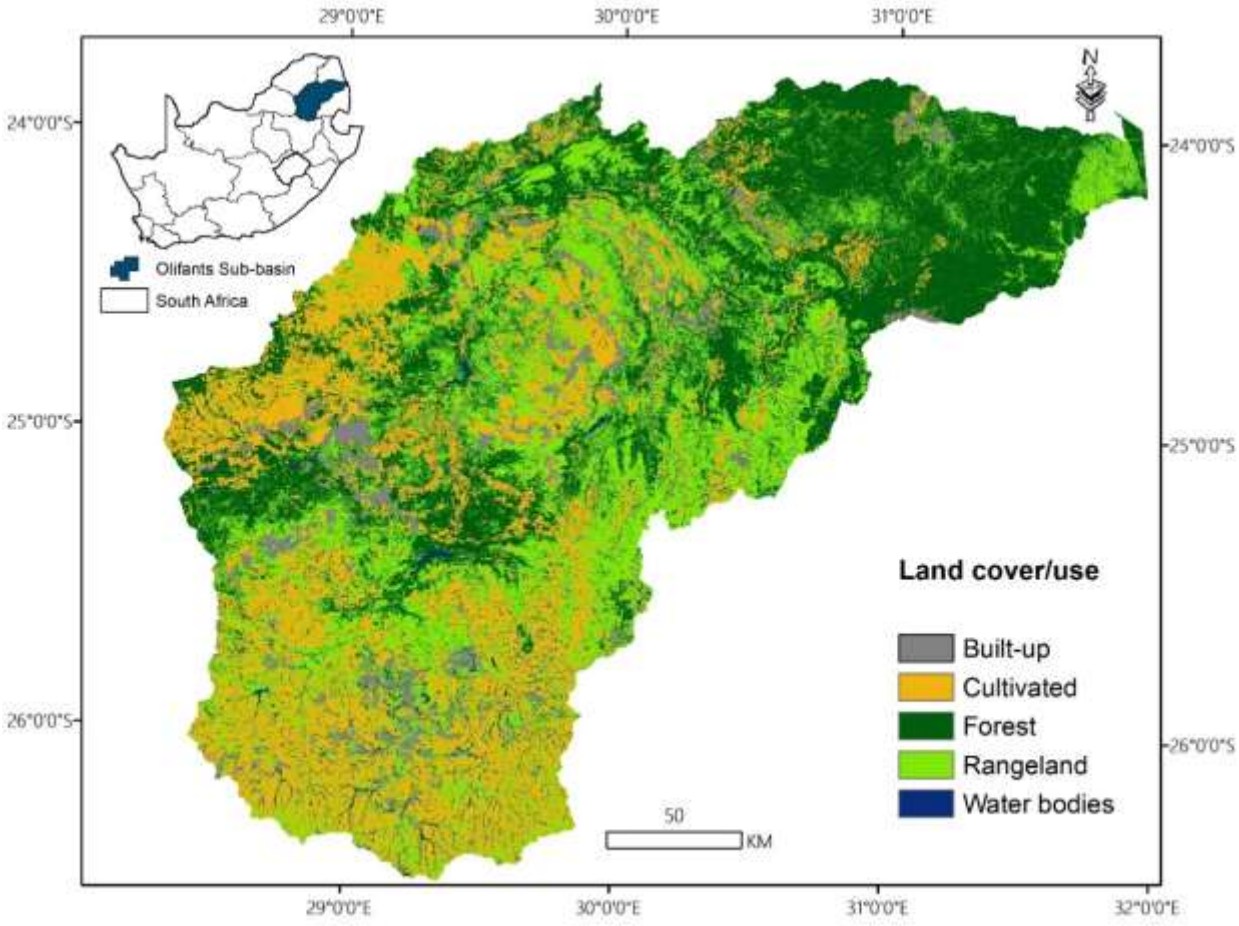


Figure 3.4. Land use/Land cover of the Olifants Sub-basin.

The total population of the Olifants Sub-basin is estimated to be about 4.2 million people. As a result, it has been subjected to extensive anthropogenic alterations and interventions. There are about 37 or more human-made major dams and 134 minor dams in the basin. Major dams have a reservoir capacity that exceeds the reservoir capacity of 2000000 m^3. In contrast, dams dubbed minor has a reservoir capacity between 100000 and 2 000000 m^3 (McCartney, 2003). All the Dams in the Sub-basin impound the total storage of 1410 000000 m^3. The dams were built to fulfil many functions. Some were constructed solely for irrigation, and multipurpose dams supply water to industries and households for domestic use. The primary water consumers are irrigation, thermal power production (i.e., cooling water), and urban water supply (McCartney, 2003). Constructing dams spiked around the 1960s, continued well into the 1980s and slightly declined in the 1990s (McCartney, 2003).

Despite the high level of water development in the Sub-basin, poverty is prevalent among the population of 2.5 million inhabitants based on a 2001 census, currently estimated at 4.2 million as of 2019. A significant percentage of households in the Sub-basin has no

income and depends on social grants. This results in high demand and competition for natural resources such as land, water, and energy for subsistence and income generation, thus increasing land degradation risk. In recent years, the Sub-basin human population has increased tremendously, leading to the conversion of formerly cultivated areas into agricultural lands. The Olifants Sub-basin is home to different economic activities that are important to the South African economy. Such as small and large-scale mining industries, commercial and smallholder agricultural farms, and tourism. These activities consume water for irrigation and the well-known Kruger National Park (Nkhonjera, 2017).

3.1.5 WATER RESOURCES AND WATER USE

Water resources in the Olifants Sub-basins are highly variable and under high demand. Water users include agriculture, forestry, mining, power generation, urban and rural communities, ecosystem flows, and tourism. All the Sub-basin water users consume about 900 Mm of surface water, and 100Mm^3 of groundwater is consumed annually (McCartney *et al*., 2004). On average, the annual consumption per person living within the Sub-basin is over 290 000 litres (McCartney *et al*., 2004). The water consumption in the sub-basin is probably higher than this with increased human population and economic development. The primary water consumers are irrigation, thermal power production, urban water supply, mining, and industrial use (McCartney *et al*., 2004).

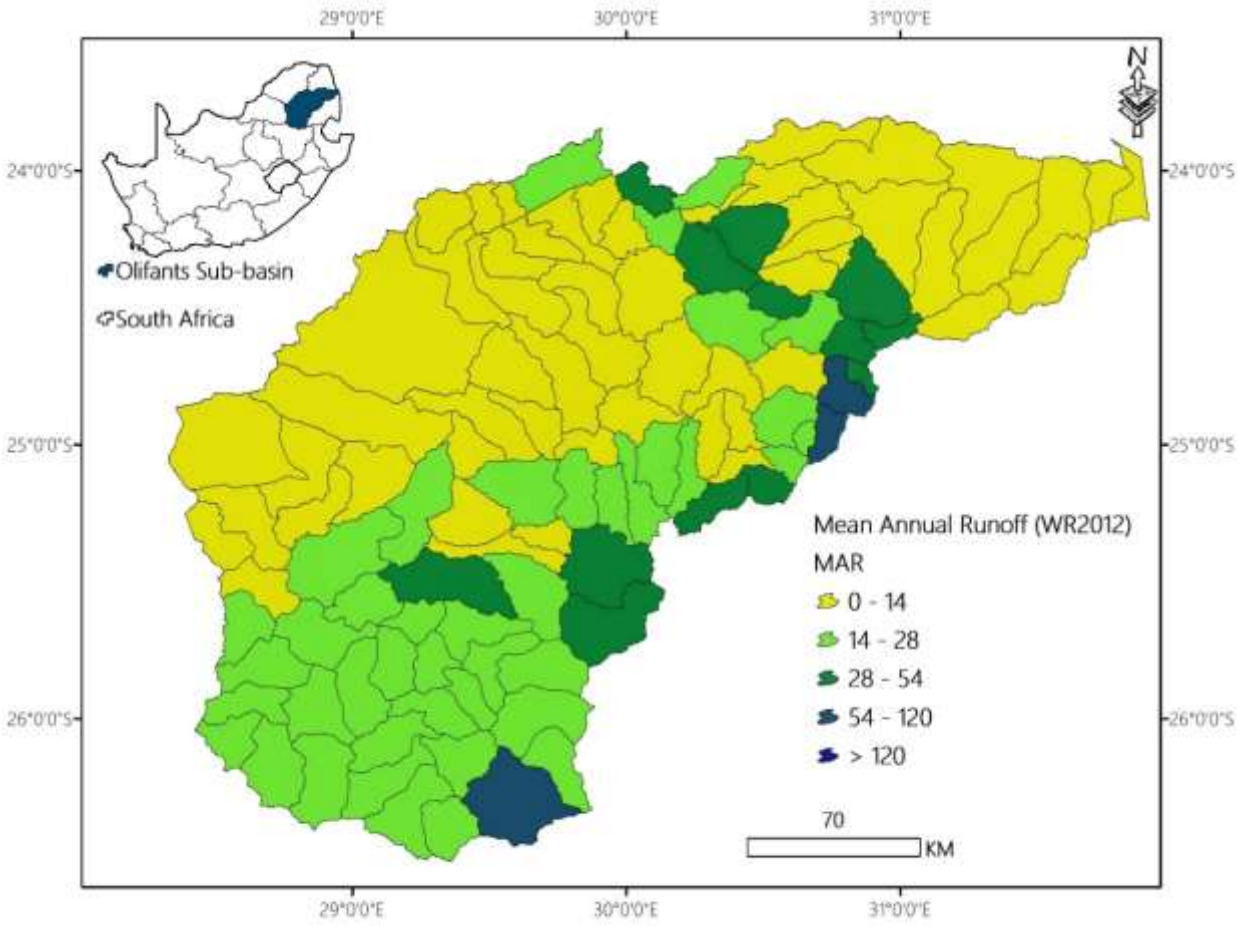


Figure 3.5. The mean annual runoff of the Olifants Sub-basin (WR2012).

Water used for power production comes from Sub-basin inter transfers. It is usually lost through evaporation, which has little impact on the hydrology of the Sub-basin. However, it may change the flow regime of the headwaters (McCartney *et al.*, 2004). The inter-basin water transfers come from the Vaal, Komati, Usutu, and Great Letaba Rivers. The expansion of agriculture and industry in the Olifants Sub-basin has been based on the available water resources. The different water consumers in the Sub-basin led to intense competition for water resources amongst small rural users, urban users, mines, industries, large-scale agriculture, forestry, and eco-tourism (Van Koppen, 2008). The Olifants River has been transformed into a complex system for harvesting, storing, transporting, and controlling water resources. The mean annual runoff (MAR) of the Olifants Sub-basin is 1235 Mm^3 and *Figure 3.5* shows how it is distributed among the quaternary catchments.

3.2 Methodology

This section details the datasets, tools, and steps used to achieve the objectives of this study. Data collection and collation took a significant amount of time. Four subsections, including model selection, data inventory, data processing, and selected SLM practices, make up this section.

3.2.1 Model Selection

Models play a significant role in assessing or predicting the impacts of sustainable land management (SLM) practices and technologies before implementation. The Soil and Water Assessment Tool – SWAT (Arnold *et al.*, 2012) is one of the most widely used watershed-scale simulation tools. It helps address watershed questions and inform catchment management worldwide. The SWAT model simulates hydrological processes using soil properties information, land use, topography, and climatic data with a relatively high spatial resolution (Mwangi *et al.*, 2015). The model is physically based. It represents the large-scale spatial heterogeneity of soil, land use, and management practices. The initial step is the topographic discretisation, which is done by dividing the catchment into sub-catchments based on a defined threshold area. For the second step, the model uses land use, slope, and soil attribute to divide the Sub-basins into unique hydrological response units (HRU). HRU refers to an area characterised by a unique combination of land cover, slope, and soil type (Neitsch *et al.*, 2011).

The SWAT model uses the water balance equation (equation 1) to drive all hydrological processes (Arnold *et al.*, 2012). Processes such as precipitation, surface runoff, infiltration, evapotranspiration, lateral flow, percolation, and groundwater flow. The equation is as follows (Neitsch *et al.*, 2011):

$$SW_t = SW_0 + \sum_{i=1}^{i=t}(R_{day} - Q_{surf} - E_a - W_{perc} - Q_{gw}) \quad \text{Equation 1}$$

Where: SW_t, SW_0 are respectively, final and initial soil water content (mm/d); t is the time (day); R_{day} is the precipitation (mm/d); Q_{surf} is the surface runoff (mm/d); E_a is the evapotranspiration (mm/d); W_{perc} is the percolation (mm/d); Q_{gw} is the return flow (mm/d).

The SWAT model calculates surface runoff using the Soil Conservation Service (SCS) curve number method for each HRU and then routes to obtain total runoff for the watershed. This approach increases the accuracy of load predictions and ensures a much better

physical description of the water balance (Neitsch *et al*., 2011). Channel routing is calculated using the Muskingum method. In contrast, the time of concentration is estimated using Manning's formula, which considers both the overland and channel flow. Neitsch *et al*. (2011) note that erosion and sediment yield is calculated for each HRU using the Modified Universal Soil Loss Equation (MUSLE).

In contrast to the Universal Soil Equation, which uses rainfall as an indicator or erosive energy, the MUSLE uses the runoff amount to simulate erosion and sediment yield (Neitsch *et al*., 2011). The MUSLE equation produces a higher prediction accuracy since it does not need a sediment delivery ratio and estimates sediment yields for a single storm (Neitsch *et al*., 2011). Potential evapotranspiration is calculated using the Penman-Monteith equation, which requires daily mean temperature, wind speed, relative humidity, and solar radiation as input variables. SWAT also simulates hydrological processes such as canopy storage, infiltration, surface runoff, and the movement and transportation of organic particles such as nitrogen and phosphorus, pesticides and sediments in the watershed (Abbaspour *et al*., 2015).

Mwangi *et al*. (2015) assert that the SWAT model is a fundamental tool to predict the impacts of various conservation measures before implementation. Such a notion forms the basis behind the selection of the SWAT model for this study. Moreover, the model was selected because it is open to access and user-friendly in handling input data. It is physically based, thus can simulate actual land management practices. It also has excellent user support and can predict changes in water and sediment yield due to a change in land management practices, which were of interest in this study. According to Gassman *et al*. (2014), the SWAT model is comprehensive and flexible in addressing water resources problems. It also has networking channels such as training workshops and ongoing international conferences, detailed online documentation, and software support.

The model has also been successfully applied even in data-poor regions of the world (Schuol *et al*., 2008). Devia *et al*. (2015) state that reasonable hydrological estimations can be obtained with the SWAT model's little direct calibration. Given that the model is successfully calibrated, it can then assess different scenarios of changing land management. With all of these fundamental attributes, the SWAT model was selected for this study to assess the potential of various proposed SLM practices and technologies.

3.2.2 DATA INVENTORY

Depending on the project's scale, the SWAT model, like any other hydrological model, requires large amounts of data. Data preparation is the most crucial aspect and the most time-consuming process in hydrological modelling. Information about the topography (digital elevation model- DEM), land cover, soil, and climate data are essential to the SWAT model. The climate data include precipitation, temperature, solar radiation, relative humidity, and wind speed. A DEM with a spatial resolution of 30-meter in an ESRI GRID format was obtained from the United States Geological Survey website. The spatial resolution of DEM has significant implications for the accuracy of model outputs. Buakhao and Kanrang (2016) assert that the DEM accuracy must be put under a microscope to ensure model success. Recommended DEM spatial resolution ranges from 30 to 300 meters. Thus, a 30 meter DEM was deemed satisfactory for this study.

Spatial information on the land cover was obtained from the National Department of Environmental Affairs in a raster format. The map was produced using Landsat 8 satellite imageries at 30 m resolution. The map's reference date is April 2013 to March 2014, when rainfall, climatic and streamflow data are available. All variables need to have a standard data period in a model (Mwangi *et al.*, 2015). The resolution of the land cover map is considered accurate enough to simulate sediment yield data accurately. With a scale of 1: 250 000, the digital soil layer was used in this study. Soil information is vital in hydrological modelling. However, soils vary significantly over short distances, which makes it challenging to obtain accurate soil data. Other GIS datasets such as the shapefiles of the study area boundary and rivers within the Sub-basin were obtained from the South African Water Resource Centre.

There are more than 46 rainfall stations within the study area. However, only 31 of these were used in this study. These stations were selected based on data availability, having less than 15 per cent missing data and data from 1989 to 2013. Of these, recorded rainfall data from 10 stations were obtained from the Department of Water and Sanitation (DWS). The remaining rainfall data from 21 stations data were obtained from the South Africa Weather Services (SAWS). The rainfall data is collected using an automatic weather station and automatic rainfall stations. More information about the data and the different gauging stations' spatial distribution are presented in *Table 3.3* and *Figure 3.6*, respectively.

Table 3.3. Summary of weather and climate stations from DWS, SAWS and CFRS.

Station Name/ ID	Elevation	Start year	End year	Data length	Missing (per cent)
239309	422	1979	2014	36	0.00
245297	809	1979	2014	36	0.00
245309	699	1979	2014	36	0.00
248300	1198	1979	2014	36	0.00
248303	1338	1979	2014	36	0.00
251284	1068	1979	2014	36	0.00
251300	1867	1979	2014	36	0.00
254291	1483	1979	2014	36	0.00
258294	1541	1979	2014	36	0.00
261297	1642	1979	2014	36	0.00
B2E001	N/A	1968	2017	49	4.12
B3E003	N/A	1936	2017	81	2.00
B3E004	N/A	1966	2017	51	5.00
B3E007	N/A	1981	2017	36	6.00
B4E003	N/A	1972	2017	45	6.00
B4E004	N/A	1972	2017	45	6.00
B6E001	N/A	1956	2017	61	7.00
B6E003	N/A	1972	2017	45	8.00
B7E003	N/A	1964	2017	53	9.00
Bankfontein	1525	1904	2017	113	0.69
Blyde	1469	1904	2017	113	8.36
Delmas	1593	1903	2017	114	1.16
Goedehoop	1570	1904	2017	113	0.29
Graskop	1436	1989	2017	29	6.00
Hendriena	1615	1904	2017	113	0.33
Houtboschrand	242	1904	2017	113	0.00
Ingwelala	321	1977	2017	40	0.00
Kingfisherspruit	448	1904	2017	113	0.00
Kriel	1555	1905	2017	112	1.10
Mariepskop	1325	1905	2017	113	0.06
Middelburg	1455	1905	2017	113	4.18
Morgenzon	1633	1905	2017	113	0.00
Oudestad	953	1980	2017	38	2.88
Salique	918	1904	2017	113	0.10
Sundra	1680	1894	2017	124	1.32
Vandyksdrift	1522	1904	2017	113	1.90
Welgevonden	1540	1904	2017	113	0.30
Witbank Colliery	1524	1904	2017	113	0.45
Witbank Strehla	1583	1904	2017	113	0.03

The observed daily datasets on wind speed, solar radiation, relative humidity, and temperature required as input SWAT model were obtained from the SAWS and Climate Forecast System Reanalysis (CFSR) database. There are seven weather stations within the Olifants Sub-basin, three of which do not fit the simulation period (1995-2017) and have less than 20 years. As a result, the three stations were not considered in the analysis. The

remaining four stations have between 23 to 39 years of recorded data with missing data percentages ranging from 3 per cent to 19 per cent.

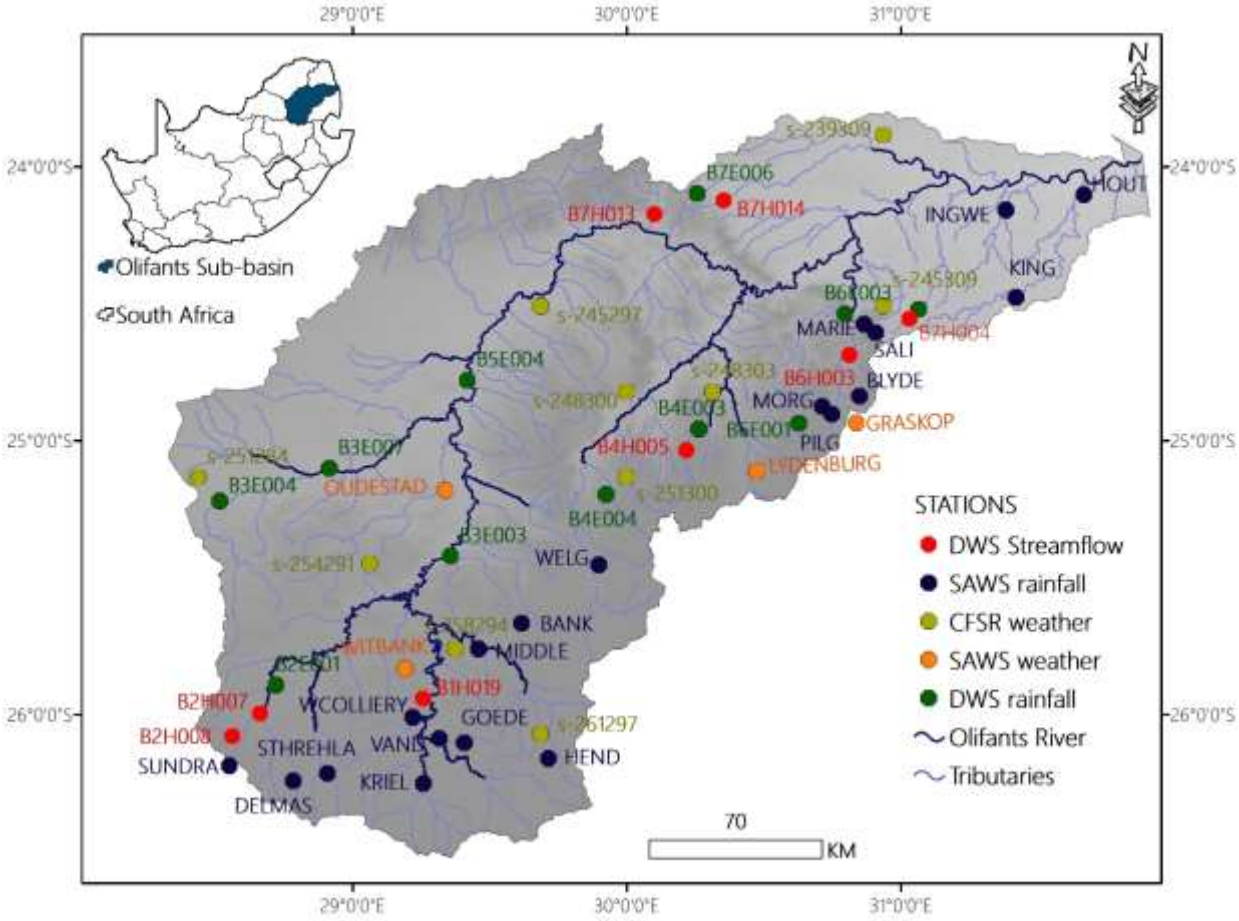


Figure 3.6. The spatial distribution of weather and flow gauging stations used in this study.

Streamflow data is essential in the SWAT model since it plays a significant role in the calibration and validation process. Ninety-nine gauging stations fall within and outside (nearby) the study area. Twenty-one stations are entirely within the study area boundary and have less than 15 per cent missing data. However, only eight stations were used in this study. Namely B1H019, B2H007, B2H008, B4H005, B6H003, B7H004, B7H013 and B7H014. These stations had uninterrupted natural flow since there are no dams in their upstream or within the catchments that they drain. These stations are likely to represent the model outputs since the reservoirs/dams within the Olifants Sub-basin were not simulated. According to a Mann-Kendell trend analysis conducted on these stations, there is a decreasing trend in annual average and maximum flow. This is supported by the reported declining rainfall trends based on Climdex analysis. The spatial distribution of the selected stations is presented in *Figure 3.6*.

Sediment load data in the study area is not measured due to different challenges such as remote, logistic constraints, and insufficient human resources (Gyamfi *et al.*, 2016). However, the DWS measures water quality variables in different sampling sites on an irregular basis. Total suspended solids (TSS) collected from a sample point (sampling ID: 90432) within the catchment (B12C) with no major dams were used. The daily TSS data was used to estimate sediment load by multiplying it with daily flow data from the same catchment. *Table 3.4* presents more information regarding the spatial resolution and sources of all the datasets used in this study.

Table 3.4. The sources of hydro-meteorological and physiographic data used in this study.

Dataset	Description	Source
Elevation	SRTM[a] digital elevation mode (30 m)	US Geological Survey
Land use/cover	Land cover map (30 m)	Department of Environmental Affairs
Soil types	Regional soil classification (1: 250 000)	Agricultural Research Council
Rainfall	Daily rainfall (mm)	DWS, SAWS
Temperature	Daily Min-Max temperature (˚C)	SAWS
Solar Radiation	Daily solar radiation (MJ/m^2)	SAWS, CFRS
Wind	Daily Wind data (m/s)	SAWS
Relative humidity	Daily humidity (per cent)	SAWS
Streamflow	Flow data (m^3/s)	SAWS
Sediment	Total suspended solids (mg/L)	DWS

[a] *Shuttle Radar Topography Mission*

3.2.3 Data processing

The input datasets in SWAT are usually required based on specific formats, from the climatic to GIS data. The spatial datasets and database files were prepared and pre-processed as per ArcSWAT requirements before simulation. Microsoft Excel customised with macros, UltraEdit, and pcpSTAT was used to process input data. ArcMap was used for all GIS data. All GIS datasets were clipped to the study area extent and projected using WSG_1984_UTM_Zone_35S spatial reference. Latitude and longitude coordinates for streamflow gauges were transformed into projected coordinates (Xpr and Ypr) to be simulated into the model as a database table. Missing data values for the weather files were replaced by -99.0 as per SWAT requirements. Weather parameters were calculated using the pcpSTAT program, and the Pivot Table function in Microsoft Excel (Table *3.5*).

Table 3.5. The description of the variables in a weather generator input file.

Variable Name	Definition
PCP_MM	Average or mean total monthly precipitation (mm H_2O)
PCPSTD	The standard deviation for daily precipitation in a month (mm H_2O/day)
PCPSKW	The skew coefficient for daily precipitation in a month
PR_W1	The probability of a wet day following a dry day in the month
PR_W2	The probability of a wet day following a wet day in the month
PCPD	An average number of days of precipitation in a month
RAINHHMX	Maximum 0.5-hour rainfall in the entire period of record for the month (mm H_2O)
RAIN_YRS	The number of years of maximum monthly 0.5 h rainfall data used to define values for RAINHHMX
SOLARAV	Average daily solar radiation for a month (MJ/m^2/day)
TNPMX	Average or mean daily maximum air temperature for the month (˚C)
TMPSTDMX	The standard deviation for daily maximum air temperature for the month (˚C)
TMPMN	Average or mean daily minimum air temperature for the month (˚C)
TMPSTDMN	The standard deviation for daily minimum air temperature for a month (˚C)
DEWPT	Average daily dew point temperature for each month (˚C)

Different soil classes were defined in the user's soil database using data from the Agricultural Research Institute: Institute of Soil, Climate and Water (ARC-ISCW) and the ACRU model datasets. The land types memoirs were used to fill in soil information for the dominant soil series within the land type coupled with the FAO's soil database. The soil information such as depth and texture from the memoirs was individually added to the SWAT2012 database. The model spatially incorporates the values of different soil properties from the SWAT2012 database into the model. Available water moisture and the K-factor was obtained from the ACRU soils dataset for South Africa. In contrast, bulk density, saturated hydraulic conductivity (SCS), and soil albedo values as shown in Table *3.2*, were obtained from Schulze (1995).

3.2.4 Selection of SLM practices and technologies

SLM practices and technologies in this study were selected based on field visits, stakeholder engagements, and existing South African literature on SLM. The project team working on the GEF5-SLM project visited some of the degraded landscapes within the Olifants Sub-basin (*Figure 3.7*). The purpose of the field visit was to enable the project team to get exposure to the communities severely affected by land degradation and engage with the local workers and community members. The field visit essentially allowed

for the project team to reconsider who are the crucial stakeholders in the GEF5-SLM project.



Figure 3.7. Degraded areas that were observed during the field visit.

The research team conducted stakeholder engagement workshops to identify key stakeholders, get their perceptions of land degradation issues in their communities and develop a community vision for sustainable land use (*Figure 3.8*). The stakeholder engagements allowed communities to recommend SLM interventions that can address land degradation and identify the actors who can support these interventions held. A participatory approach was used during the stakeholder engagement workshops to which allowed for the collective inquiry to identify the problem (land degradation) and solutions based on experience, a reflection of social history and scientific input. Some of the selected and proposed SLM practices and technologies include contour farming, filter strips, parallel terraces and agroforestry.



Figure 3.8. Stakeholder engagement workshops held in the Lepellane catchment (Sekhukhune District Municipality, Limpopo).

3.3 Model set up

The ArcSWAT 2012 interface was used to set up and run the model. The ArcSWAT updated version 2012.10_4.21, released on the 9th of October 2018, was downloaded from the official SWAT website. After the software was downloaded, the Spatial Analyst and ArcSWAT extensions were enabled under tools to avail the SWAT menu. The SWAT menu encompasses items that control the setup and management of the SWAT project. The menu includes the SWAT Project Setup, Watershed Delineator, HRU Analysis, Write Input Table, Edit SWAT input, and SWAT Simulation menus. There are other items or steps under each menu. As each procedure is completed, the next menu becomes available subsequently. Under the SWAT Project Setup, the New SWAT Project command was

selected to create a New SWAT project directory structure. All prepared input files were saved in the defined project directory; this allowed the model to access the datasets where necessary automatically.

3.3.1
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3.3.1 Watershed delineation

This menu contains all the commands used to perform sub-basin delineation. Watershed delineation allows the user to delineate watersheds using either an automatic procedure based on the Digital Elevation Model (DEM) or using pre-existing streams and watersheds. The automatic watershed delineation tool subdivides a watershed into Sub-basins and generates the stream network based on the DEM. A DEM was used to define the streamflow and direction automatically. The existing river layer in a vector format was used to divide the Olifants Sub-basin into hydrologically connected sub-watersheds and accurately help the DEM predict stream location. The threshold drainage area was set to 26980 hectares to discretise the Olifants Sub-basin into 122 Sub-basins. Observed streamflow gauges were added to the sub-basin outlets using the add Sub-basins outlet by table option under the Outlet and Inlet Definition tab. Historical observed streamflow data are necessary for comparing the observed and simulated flows during the calibration process. Sub-basins parameters such as the channel length and average slope of the channel were calculated.

3.3.2 Modelling: calibration, and validation

The model was run for the period 1995 to 2017 at a monthly timescale. Two to three years is considered satisfactory for a warm-up period (Abbaspour, 2013). The warm period is sufficient to allow the model to establish the initial conditions such as soil water content. The year 1995, 1996, and 1997 were used as a warm-up period to allow the model to get a sense of the conditions that are typical of normal running conditions such as rainfall and rainfall. As a result, the SWAT model does not produce output files for this period.

Hydrological models are routinely calibrated, validated, and analysed for sensitivity and uncertainty (Abbaspour *et al*., 2011). Necessitated by the fact that models are not an exact representation of the real-world, poor quality of input data and issues of representing actual processes that occur in nature. Some of the observed streamflow data were used for model calibration (1998 – 2007), and others for validation (2008 – 2017). Based on the modelling period and excluding the warm-up years (1998-2017), both calibration and

validation monthly averages were supposed to be 120 in total. However, due to missing data values, some stations had data values less than the supposedly total (120). Models should be calibrated and validated before they are used in decision-making for water resources management. Calibration is an essential step in modelling. It aims to minimise differences between the model output and measured data (Abbaspour *et al*., 2017). It is, therefore, a parameter values fine-tuning procedure to optimise model performance based on specific predefined criteria (Jajarmizadeh *et al*., 2017).

The Sequential Uncertainty Fitting – SUFI-2 algorithm (Abbaspour, 2013) within the SWAT-CUP environment was used to calibrate, analyse parameter sensitivity, and validate the model. The SWAT-CUP interface provides a link between the input/output of a calibration program and the SWAT model (Abbaspour, 2013). The SUFI-2 algorithm determines the uncertainties that arise from model input, model structure, selected parameters, and observed data through sequential and fitting processes (Brighenti *et al*., 2019). The 95 per cent prediction uncertainty (95PPU) graphically represents the model uncertainties (Abbaspour *et al*., 2017). The 95PPU is calculated at 2.5 and 97.5 intervals of the cumulative distribution of an output variable acquired through Latin hypercube sampling (Abbaspour *et al*., 2015). Uncertainty analysis plays a vital role in assessing the calibrated model's strength (Abbaspour *et al*., 2015).

The *P-factor* and *R- factor* statistics are used to determine the relationship between the simulated (represented by 95PPU) and the observed data such as discharge. *P-factor* quantifies the goodness of fit between the simulated results expressed as 95PPU and measured data (Abbaspour *et al*., 2015). *P-factor* ranges from 0 to 1, with values greater than 0.7 regarded as adequate. The *R-factor* depicts the thickness of the 95PPU band given by the ratio of the average width of the 95PPU band and the standard deviation of the observed variable (Abbaspour, 2013). *R-factor* values range from zero to infinity, and values less than 1.5 are deemed adequate, depending on the project's extent and the quality of the input and calibrating data (Abbaspour *et al*., 2015). A *P-factor* of 1 and an *R-factor* of zero in a simulation indicates a perfect match between the simulated and measured data (Abbaspour, 2013).

When calibrating the SWAT model using variables such as discharge and sediment yield, it is best to first calibrate discharge. This procedure is suggested by Abbaspour (2013) due to discharge being the primary controlling variable in a hydrological system. The approach is employed in this study and as a result, the model was first calibrated for streamflow at a monthly timescale using parameters that are sensitive to streamflow. Table *3.6* presents the parameters used in this study. Before the parameters sensitive to

sediment load were added, the calibration model was run for three iterations (Table *3.6*). At each iteration, the ranges of the selected parameters ranges were updated.

Table 3.6. The selected SWAT model discharge and sediment parameters for calibration and validation.

Parameter	Description of parameter
Discharge	
CN2	SCS runoff Curve number
ALPHA_BF	Baseflow Alpha factor (days)
GW_DELAY	Groundwater delay (days)
GWQMN	Threshold depth of water in the shallow aquifer required for return flow to occur (mm)
GW_REVAP	Groundwater "revap" coefficient
ESCO	Plant uptake compensation factor
CH_K2	Effective hydraulic conductivity in main channel alluvium (mm/hr)
ALPHA_BNK	Baseflow alpha factor for bank storage
SOL_AWC	Available water capacity of the soil layer
SOL_K	Saturated hydraulic conductivity
REVAPMN	Threshold depth of water in the shallow aquifer for "revap" to occur (mm)
RCHRG_DP	Deep aquifer percolation fraction
SURLAG	Surface Runoff lag time (days)
SOL_BD	Moist bulk density (g/cm3)
CH_N2	Manning's "n" value for the main channel
CANMX	Maximum canopy storage (mm H2O)
Sediment	
PRF_BSN	Peak rate adjustment factor for sediment routing in the main channel
SPEXP	Exponent parameter for calculating sediment re-entered in channel sediment routing
SPCON	Linear parameter for calculating the channel sediment routing
CH_COV2	Channel cover factor
USLE_P	USLE equation support parameter
USLE_C	A minimum value of USLE C factor applicable to the land cover/plant
USLE_K	Soil erodibility factor in USLE
CH_EROD	Channel erodibility factor

Sensitivity analysis was conducted to determine the parameters that significantly influence the model behaviour. Abbaspour *et al.* (2017) assert that sensitivity analysis plays a significant role in identifying the most significant parameters representing the study area's hydrological process. It helps to exclude parameters that are not sensitive in the calibration process. There are two types of sensitivity analysis, namely one-at-a-time and

global sensitivity analysis. The latter was deemed adequate for this study since it is more reliable than the one-at-a-time sensitivity analysis (Abbaspour *et al.*, 2017).

Global sensitivity changes all parameters simultaneously to identify their effect on the model outputs or the objective function (Abbaspour *et al.*, 2017). The *t-test* and *p-value* were then used to determine the relative significance of each parameter. Abbaspour (2013) indicates that the sensitivities are based on the average change in the objective function due to parameter changes. The objective function used in this study is the renowned Nash-Sutcliffe-NS (Nash and Sutcliffe, 1970). The NS objective function is the most widely used objective function in hydrological modelling and SWAT model application (Mwangi *et al.*, 2015; Gyamfi *et al.*, 2016a; Gyamfi *et al.*, 2016b; Brighenti *et al.*, 2019). It measures how well the residual variance's relative magnitude corresponds with the measured data variance (Nash and Sutcliffe, 1970). The NS is represented mathematically as:

$$NS = 1 - \frac{\sum_i (Q_m - Q_s)_i^2}{\sum_i (Q_{m,i} - \bar{Q}_m)^2} \qquad \text{Equation 2}$$

Where: Q is the variable such as discharge, m and s stand for measured and simulated, $\bar{Q}$ stands for the mean of the observed variable.

Sensitivity ranking of parameters was conducted based on the two statistics. The *t-test* determines the magnitude of sensitivity, whereas the *p-value* is used to determine the most sensitive parameters. A *p-value of* less than 0.05 indicates a significant relationship between the independent (predictor's value) and the dependent (response) variable; thus, the null hypothesis is rejected. In this case, the null hypothesis states that there is no change in the objective function as the parameters change. A parameter is most sensitive when it has a larger *t*-stat and smaller *p*-value respect (Abbaspour *et al.*, 2017).

After calibration, the model was then validated for flow and sediment to determine model performance based on selected indices. Validation ensures confidence in the calibration parameters and model setup (Abbaspour *et al.*, 2017). The model was validated using measured data from 2008 to 2017 without changing the final calibration parameters. Model performance indices used in conjunction with the *P- and R-factor* include the Nash-Sutcliffe (NS), Coefficient of determination (R^2), Root mean square error-observation standard deviation ratio (RSR), and Percent bias (PBIAS). These statistical indices are recommended by Moriasi *et al.* (2007) to evaluate model performance.

NS values range from -∞ to 1, with NS = 1 being the optimal value and values less or equal to 0.4 considered unsatisfactory. R^2 measures the correlation between the observed

and simulated variables and the percentage of variation in the simulated variable explained by the measured variable (Gyamfi *et al.*, 2016a). R^2 values range from 0 to 1, with values greater than 0.5 regarded acceptable. PBIAS evaluates the simulated output's average tendency to underestimates or overestimates the observed data (Gupta *et al.*, 1999). Positive and negative values indicate that the model underestimates and overestimates the observed variable, respectively, and the optimal value being zero (Ayele *et al.*, 2017). RSR is an error-index statistic that standardises the root mean square error using the observation standard deviations. RSR values vary from zero to large positive numbers, and the lower the value, the better the model fit (Moriasi *et al.*, 2007). Statistical indices used to evaluate model performance in this study, adopted from Moriasi *et al.* (2007) and Ayele *et al.* (2017), are presented in Table *3.7*.

Table 3.7. Statistical indices used to evaluate model performance in this study.

Indices	Value range	Performance classification
Coefficient of determination (R2)	$0.7 < R^2 < 1$	Very good
	$0.6 < R^2 < 0.7$	Good
	$0.5 < R^2 < 0.6$	Satisfactory
	$R^2 < 0.5$	Unsatisfactory
Nash-Sutcliffe (NS)	$0.75 < NS \leq 1$	Very good
	$0.65 < NS \leq 0.75$	Good
	$0.50 < NS \leq 0.65$	Satisfactory
	$NS \leq 0.4$	Unsatisfactory
Per cent bias (PBIAS)	$PBIAS < \pm 10$	Very good
	$\pm 10 \leq PBIAS < \pm 15$	Good
	$\pm 15 \leq PBIAS < \pm 25$	Satisfactory
	$PBIAS \geq \pm 25$	Unsatisfactory
Root mean square error-observation standard deviation ratio (RSR)	$0.00 \leq RSR \leq 0.50$	Very good
	$0.50 \leq RSR \leq 0.60$	Good
	$0.60 \leq RSR \leq 0.70$	Satisfactory
	$RSR > 0.70$	Unsatisfactory

3.4 Simulation of SLM Practices

Most hydrological modellers ignore the risk analysis step. A modeller can then quantify the problem and its impact on the environment (Abbaspour *et al.*, 2017). Abbaspour *et al.* (2017) assert that this is due to uncertainties around model outputs, the lack of supporting literature on decision-making, and a standard protocol or procedure on communicating uncertainties to decision-makers. Risk analysis can be used instead. The probability of loss is multiplied by the cost of loss, such as the probability of soil loss and the cost of soil loss

(Abbaspour *et al*., 2017). Such analysis is quantified by manipulating and propagating risk associated parameters ranges.

Once the model was successfully calibrated, the fitted parameter values were then transferred and fitted to the SWAT model. There are so many SLM practices and technologies that can be used to addressed land degradation and can be applied in the SWAT model. However, for this study, the proposed SLM best practices only include contour farming, filter strips, parallel terraces, and agroforestry. These SLM practices and technologies were selected based on field visits, stakeholder engagements and existing South African literature on SLM. Corresponding parameters in SWAT were updated to simulate the impacts of these practices. Filter strips increase infiltration while reducing run-off volume and sediment (Mwangi *et al*., 2015). The SWAT model has a FILTERW parameter in-house. The filter strip width parameter was adjusted at 5 m intervals to simulate the filter strip practice and its sensitivity on streamflow and sediment yield.

Curve number (CN) and USLE practice factor were modified to simulate contour farming impacts accordingly. The curve number accounts for increased storage and infiltration, whereas the USLE practice factor counts for decreased erosion. As a result, these parameters were reduced to simulate the positive effects of contour farming. Specifically, the CN parameter was reduced by three units as recommended by Arabi et al. (2008). The reduced PUSLE parameter value was selected based on the HRU slope (
Table 3.8). Parallel terraces reduce surface run0ff by impounding water in small depressions, decreases slope length, which then reduces peak runoff rate and sheet erosion by trapping sediments (Arabi *et al*., 2008). The SLM practice was simulated in the SWAT model by adjusting the USLE practice factor (USLE_P), slope length (SLSUBBSN), and curve number (CN). The SLSUBBSN parameter was estimated for each parallel terrace based on its type and slope of the HRU where it is installed. SLSUBBSN is given by:

$$SLSUBBSN = \frac{(A \times S + B) \times 100}{s} \qquad \text{Equation 3}$$

The s variable represents the average slope of the HRU, and A is equal to 0.21 for low rainfall in the study area. Variable B is equal to 0.3 to account for highly erodible soils and little plant cover in the selected Sub-basins catchments (Arabi et al., 2008). CN values were reduced by six units from the calibrated value to factor in the increased infiltration as in Aribi et al. (2008) and Mwangi et al. (2015). USLE_P values for terraced conditions of Type 1 were adapted from Wischmeier and Smith (1978) as shown in
Table 3.8.

Table 3.8. USLE_P parameter values for contouring, strip-cropping, and terracing (adapted from Wischmeier and Smith, 1978).

Land Slope (%)	USLE_P			
	Contour farming	Strip cropping	Terracing	
			Type 1[a]	Type2[b]
1 to 2	0.60	0.30	0.12	0.05
3 to 5	0.50	0.25	0.10	0.05
6 to 8	0.50	0.25	0.10	0.05
9 to 12	0.60	0.30	0.12	0.05
13 to 16	0.70	0.35	0.14	0.05
17 to 20	0.80	0.40	0.16	0.06
21 to 25	0.90	0.45	0.18	0.06

[a] = Graded channel sod outlet [b] = Steep backslope underground outlet

Agroforestry is the practice of planting trees with crops and livestock. It includes intercropping sparsely distributed trees with different crops, planting trees along the hedges and borders, and woodlots. It presents a challenge trying to simulate other agroforestry types in SWAT due to the model's structure (Mwangi *et al.*, 2016). For that reason, to simulate agroforestry in this study, agricultural HRUs were converted into forestry HRU to increase tree cover or woodlots. CN2 values for agricultural HRUs were converted to wood-grass combination (orchards or tree farm) values based on hydrologic conditions and the hydrological soil group of the HRU (Arnold *et al.*, 2013). More information regarding the manipulated parameters and the values used is shown in *Table 3.9*.

Table 3.9. Representation of Agroforestry, Filter Strips, Contour Farming and Parallel terraces in SWAT.

SLM practice	Function	SWAT Parameter	Range	Value when SLM implemented
Agroforestry	Change infiltration properties.	CN2	2 – 98	36 (CN2 value for woods orchard or tree farmland use)
Filter strips	Increase sediment trapping	FILTERW	0 – 30 m	Intervals of 5 m
Contour farming	Reduce surface runoff.	CN2	2 – 98	Three units down
	Reduce sediment yield.	USLE_P	0 – 1	Based on slope in *Table 3.8*
Parallel terraces	Reduce overland flow.	CN2	2 – 98	Six units down
	Reduce sheet erosion.	USLE_P	0 -1	Based on slope in *Table 3.8*
	Reduce slope length.	SLSUBBSN	10 -150	*SLSUBBSN = (A x s + B) x 100/s*

4. RESULTS AND DISCUSSION

This chapter presents the findings from all the analyses conducted to answer the stated research questions in response to study objectives. The results range from the number of sub-basins in the model set up to the percentage of change in simulated streamflow and sediment yield given the proposed SLM practices and technologies.

4.1. Definition of Hydrological Response Units: Land use, Soil, and Slope

As mentioned before, Sub-basins can further be subdivided into hydrological response units (HRU). Each HRU represents a unique combination of soil, slope, and land cover within the Sub-basin. First, the land use and soil layers were defined and reclassified to match the SWAT classification format. The slope raster layer was classified into three classes, namely flat to gently undulating (<8 per cent), rolling to hilly (8 per cent- 30 per cent), and steeply dissected to mountainous (>30 per cent). The flat to gently undulating slope has the highest area at 72%, followed by the rolling to hilly slope at 20%. Lastly, it is steeply dissected to the mountainous slope area at 8% per cent.

While the Sub-basins are delineated and located spatially, the further subdivision into HRUs is performed statistically. The subdivision is done by considering a certain percentage of the Sub-basin area, without any specific location in the Sub-basin (van Griensven *et al.*, 2006). The HRUs were defined by using the pragmatic approach. The approach creates at least one HRUs for each Sub-basin based on the given threshold values on percentage areas of land use, slope, and soil. This study used 10 per cent as the threshold value for land use, soils, and slope. If the area of the soil type, slope, or land cover type is less than ten per cent of the Sub-basin area, it is considered negligible. It is, therefore, not included in the subsequent analysis. The resulted in 1246 HRUs for the study area. The HRUs distribution report for each Sub-basin is attached in Appendix B.

4.2. Sensitivity Analysis

The results of the relative sensitivity of parameters based on the Latin Hypercube-Global analysis are as shown in *Table 4.1*. Ten parameters of the initial sixteen parameters are sensitive to streamflow based on the t-stat and p-value ($p < 0.05$). The *t*-stat determines the degree of sensitivity, and the *p*-value indicates the statistical significance of the parameter. Therefore, a parameter with a more considerable t-stat absolute value and

smaller p-value ranked as the most sensitive to streamflow. The global sensitivity analysis shows that the most sensitive streamflow parameters represent landscape properties, surface runoff, and soil properties.

Table 4.1. The ten most sensitive parameters to streamflow based on the Latin Hypercube-Global analysis. Sensitive variables are defined as those with P < 0.05 and larger t-Stat.

Rank	Parameter Name	t-Stat	P-Value
1	R__CN2.mgt	-41.44	0.00
2	R__SOL_BD(..).sol	-29.33	0.00
3	V__ESCO.hru	-15.35	0.00
4	V__ALPHA_BNK.rte	-9.89	0.00
5	R__SOL_AWC(..).sol	6.42	0.00
6	V__GW_DELAY.gw	4.77	0.00
7	R__SLSUBBSN.hru	4.30	0.00
8	V__CH_K2.rte	4.11	0.00
9	R__HRU_SLP.hru	-3.77	0.00
10	V__CANMX.hru	3.23	0.00

The SCS runoff curve number (CN2) is the most sensitive parameter on streamflow. CN2 parameter is spatially based and predicts direct runoff or infiltration from rainfall excess. Additionally, soil density (SOL_BD), evaporation compensation factor (ESCO), base flow alpha factor for bank storage (ALPHA_BNK), soil available water capacity (SOIL_AWC), groundwater delay (GW_DELAY), average slope length (SLSUBBSN), effective hydraulic conductivity in main channel alluvium (CH_K2, mm/hr), average slope steepness (HRU_SLP) and maximum canopy storage (CANMX) also showed considerable sensitivity in streamflow simulation. The high ranking of importance for CN2, ESCO, SOL_AWC, and GW_DELAY in simulating streamflow is consistent with other studies (Asharf Vaghefi *et al*., 2014; Gyamfi *et al*., 2016c; Raihana *et al*., 2019).

Parameters such as the saturated hydraulic conductivity (SOL_K); surface runoff lag time (SURLAG); baseflow alpha-factor (ALPHA_BF, days); threshold depth of water in the shallow aquifer for "revap" to occur (REVAPMN); deep aquifer percolation fraction (RCHRG_DP); groundwater "revap" coefficient (GW_REVAP); threshold depth of water in the shallow aquifer required for return flow to occur (GWQMN) and Manning's "n" value for the main channel (CH_N2) were included in the sensitivity analysis. However, they were statistically insignificant (p > 0.05), meaning that a change in their input values led to no significant streamflow change. The current study's findings conform to results from a previous study conducted in the Olifants Sub-basin (Gyamfi *et al*., 2016c). Other studies indicate that ALPHA_BF, GWQMN, REVAPMN, RCHRG_DP are sensitive to streamflow

(Memarian *et al*., 2014; Jang *et al*., 2017; Raihana *et al*., 2019). The varying results are due to spatial differences and geomorphological characteristics of the particular area under investigation.

Sensitivity analysis, carried out using nine sediment parameters, identified the five most sensitive parameters controlling the output variable (*Table 4.2*). The most sensitive parameters are a minimum value of USLE C factor applicable to land cover/plant (USLE_C); Soil erodibility factor in USLE (USLE_K); Channel cover factor (CH_COV2); USLE equation support parameter (USLE_K), and peak rate adjustment factor for sediment routing in the main channel (PRF_BSN). Most of these parameters are related to land use practices and surface runoff. Parameters CH_COV2 and USLE_C were also identified as significant sensitive parameters to sediment yield in the Sub-basin (Gyamfi *et al*., 2016a). Other studies indicate that parameters such as the linear parameter for calculating the channel sediment routing (SPCON), Exponent parameter for calculating the sediment reentered in channel sediment routing (SPEXP), USLE equation support parameter, and Channel erodibility factor (CH_EROD) are most sensitive to sediment yield (Gyamfi et al., 2016a; Gull *et al*., 2017). Spatial variability leads to varying results.

Table 4.2. The five sensitive parameters to sediment based on the Latin Hypercube-Global analysis. Sensitive variables are defined as those with P < 0.05 and larger t-Stat.

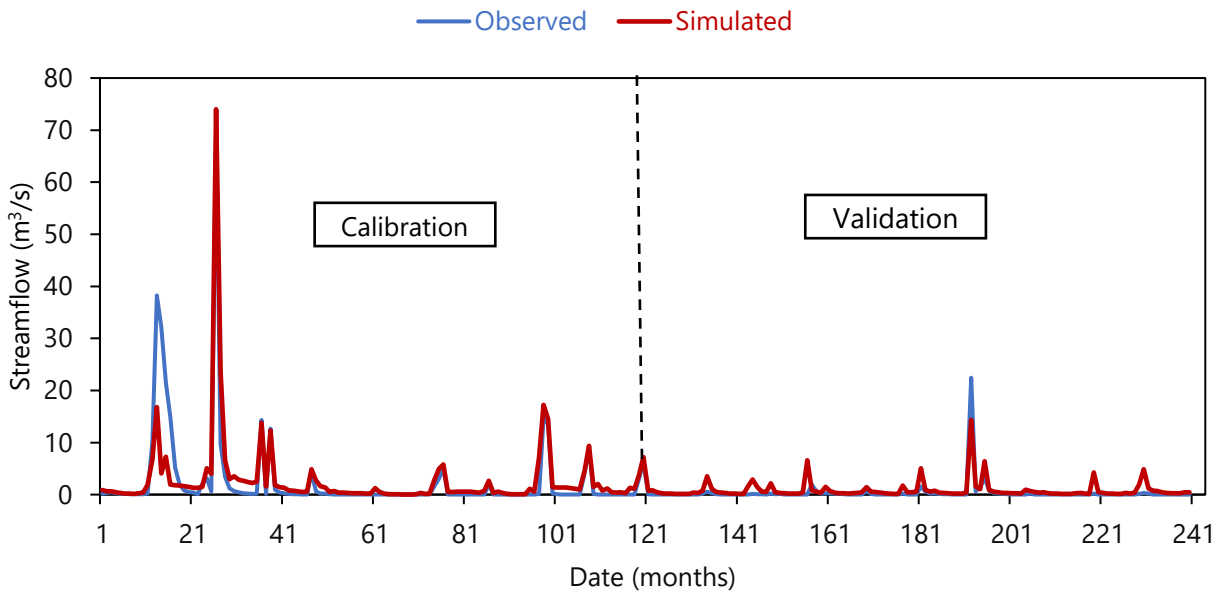
Rank	Parameter Name	t-Stat	P-Value
1	v__USLE_C.plant.dat	-44.12	0.00
2	r__USLE_K(..).sol	-18.80	0.00
3	r__CH_COV2.rte	-2.30	0.02
4	r__USLE_P.mgt	-2.25	0.03
5	v__PRF_BSN.bsn	2.23	0.03

4.3. Calibration and Validation

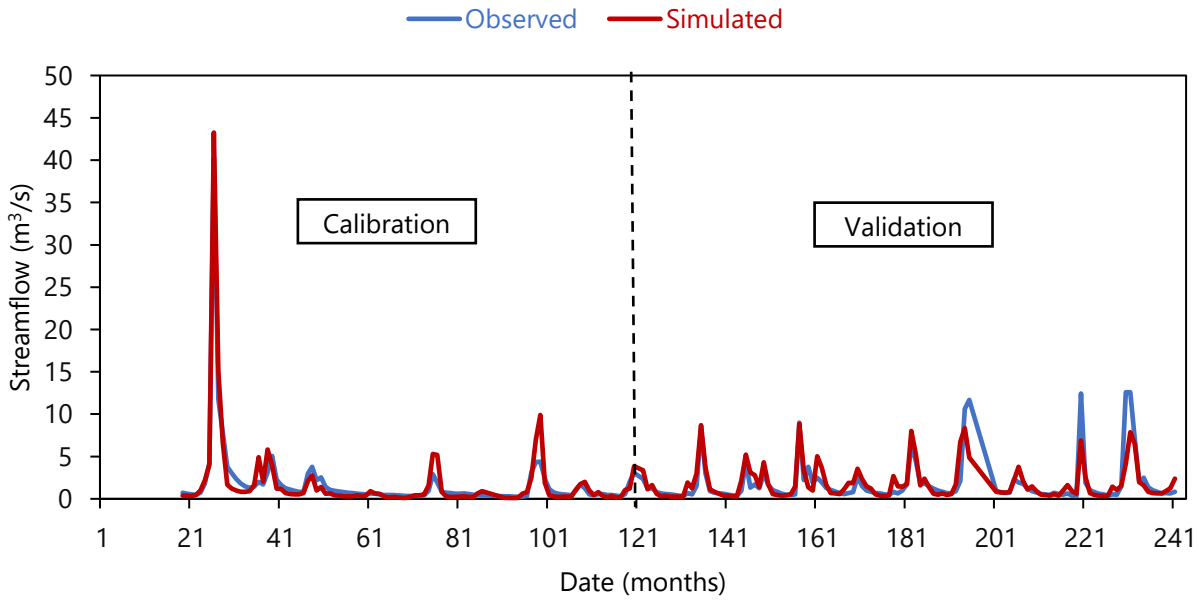
The graphical comparison of observed and simulated streamflow and sediment yield for the calibration (1998-2007) and validation (2008-2017) periods is shown in *Figure 4.*1 and *Figure 4.2*. The SWAT model has captured the observed variables (streamflow and sediment yield) satisfactorily. However, in some outlets, peak flows are underestimated (B7H014 and B7H013) and overestimated (B4H005 and B1H019) in some instances. Missed or underestimated peak flows are found in stations located in the lower part of the Olifants Sub-basin. In contrast, overestimated peak flows are in the upper part of the Sub-

basin. Such trends are likely due to intensive urban development in the lower part of the Sub-basin.

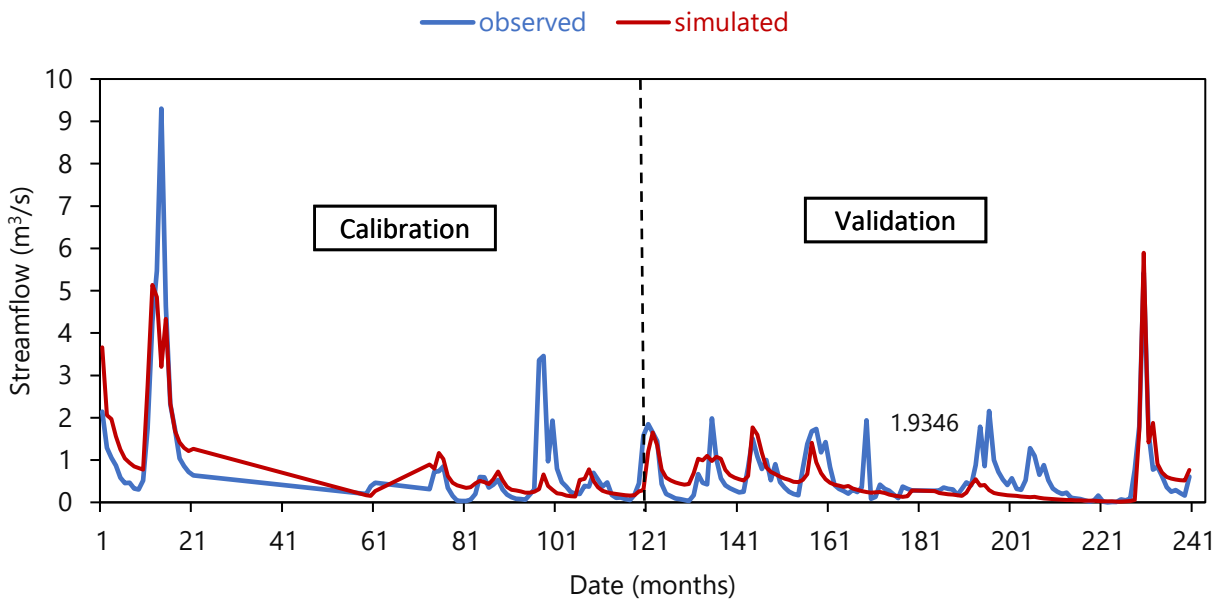
a) B72E QC/B7H014



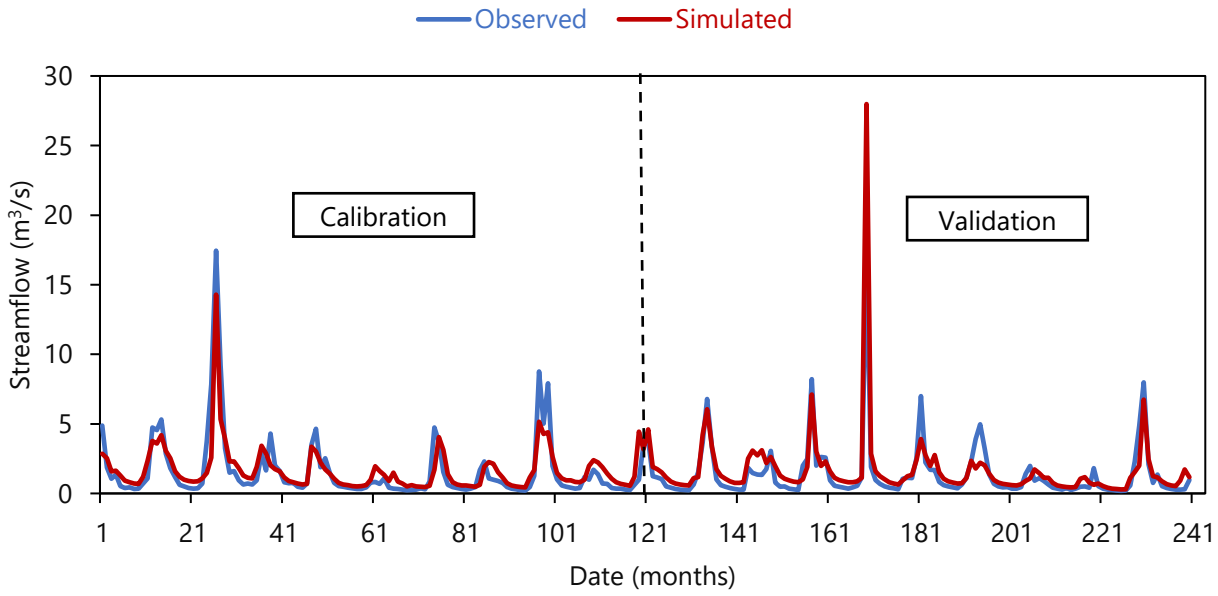
b) B71D QC/B7H013



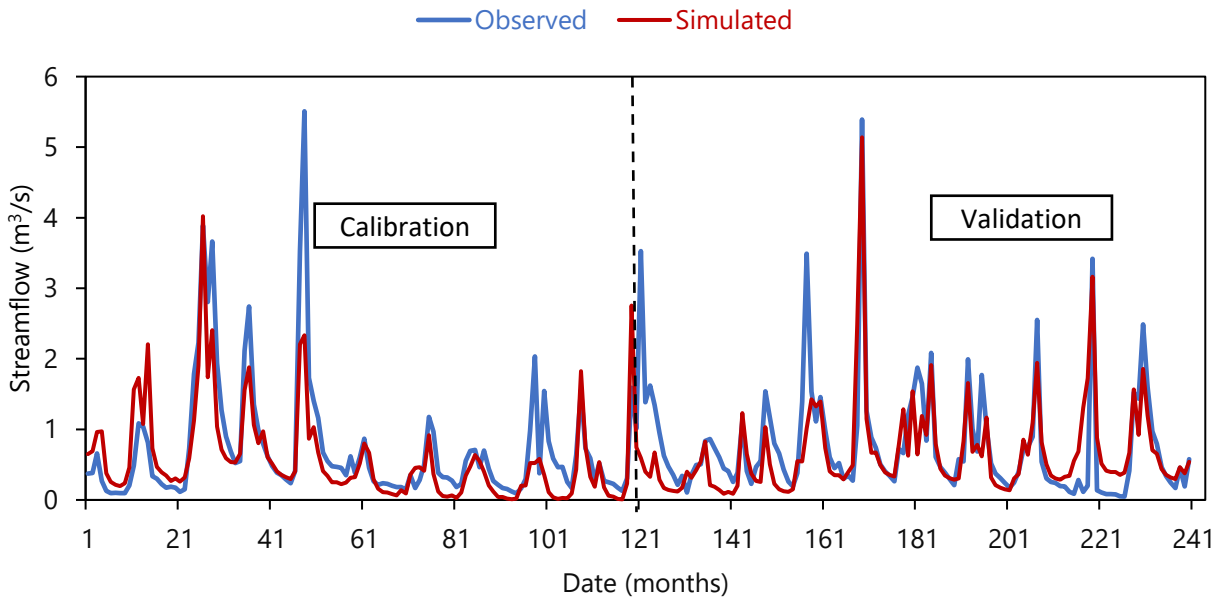
c) B73B QC/B7H004



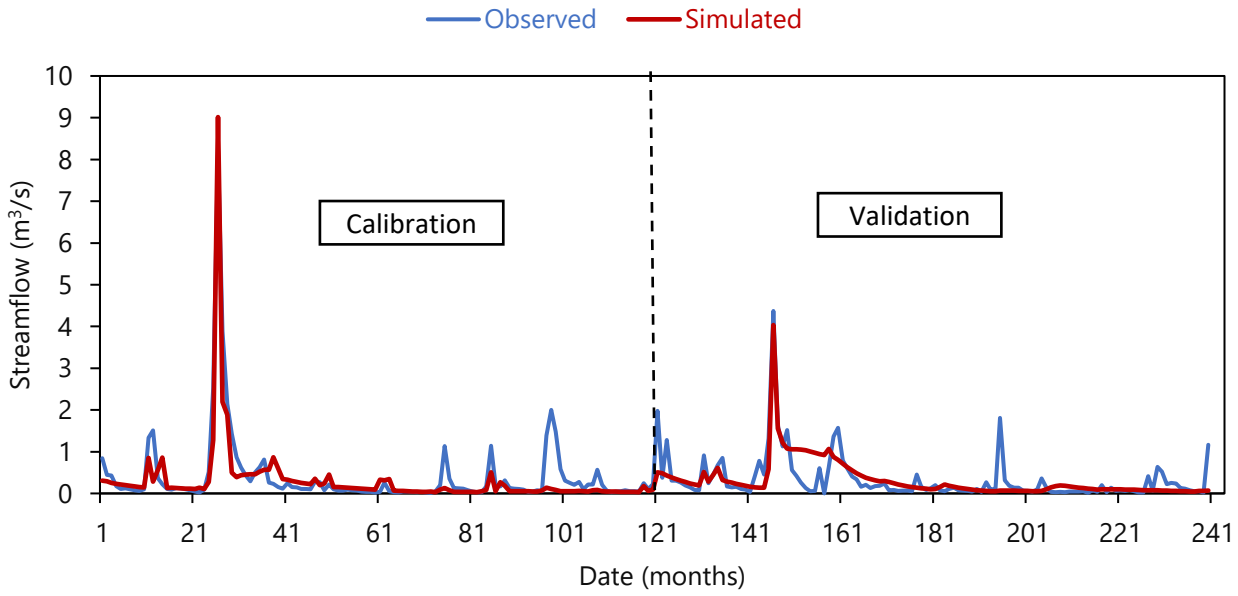
d) B60D QC/B6H003



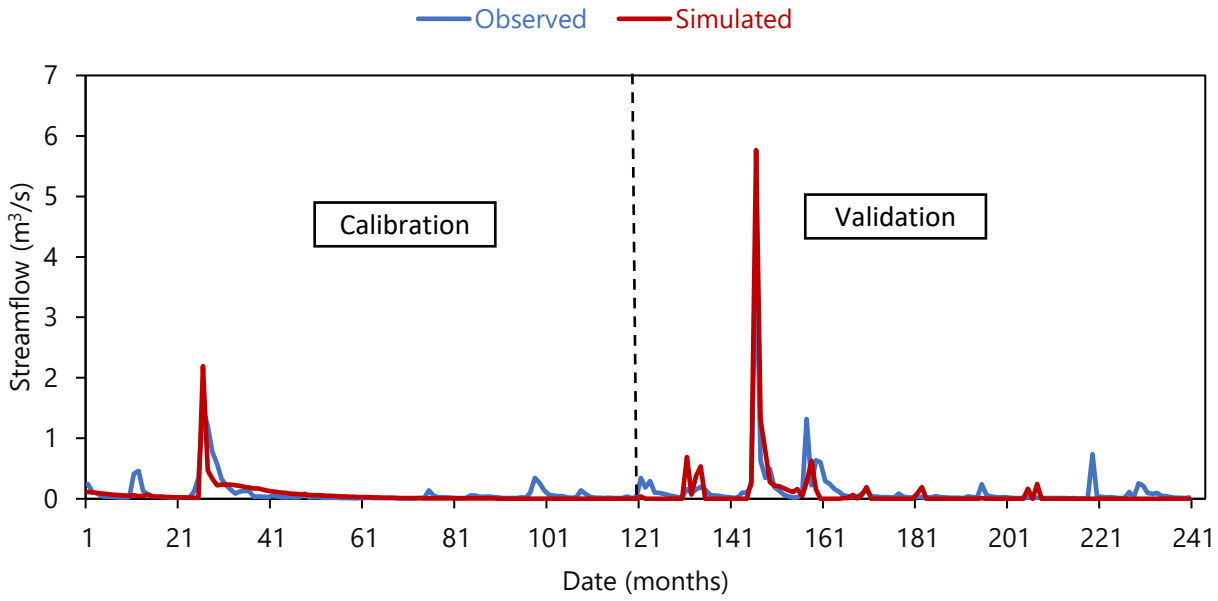
e) B42F QC/B4H005



f) B20B QC/B2H007



g) B20B QC/B2H008



h) B11G QC/B1H019

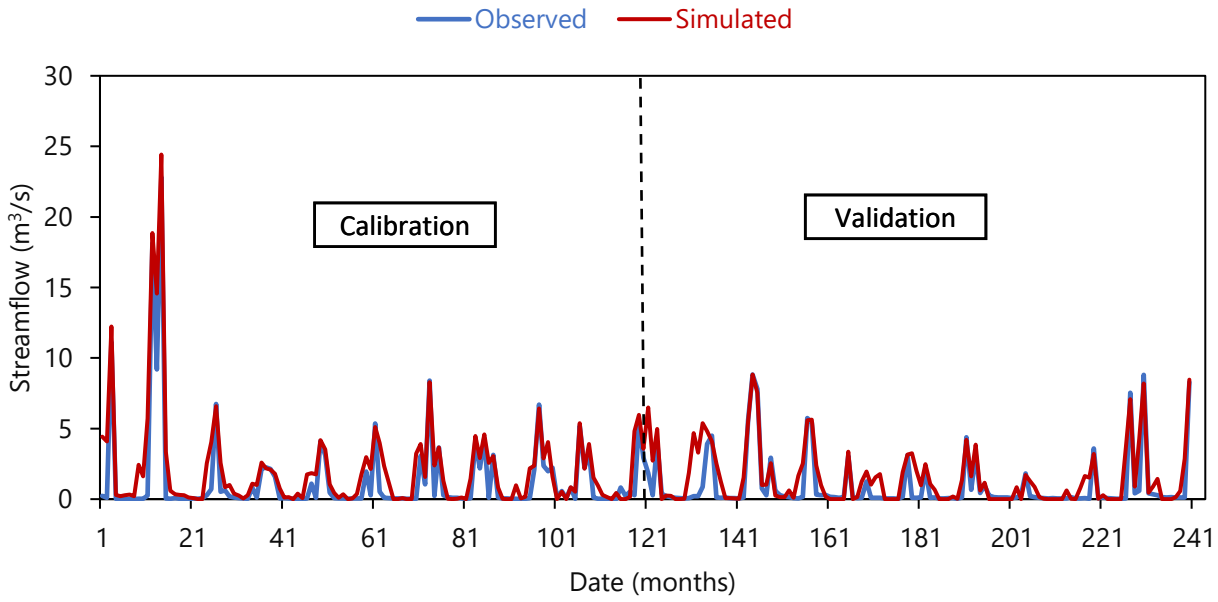


Figure 4.1. The observed and simulated monthly streamflow of the selected gauges (a-h) for calibration (1998-2007) and validation (2008-2017) periods.

a) B12C QC/Sampling ID 90432

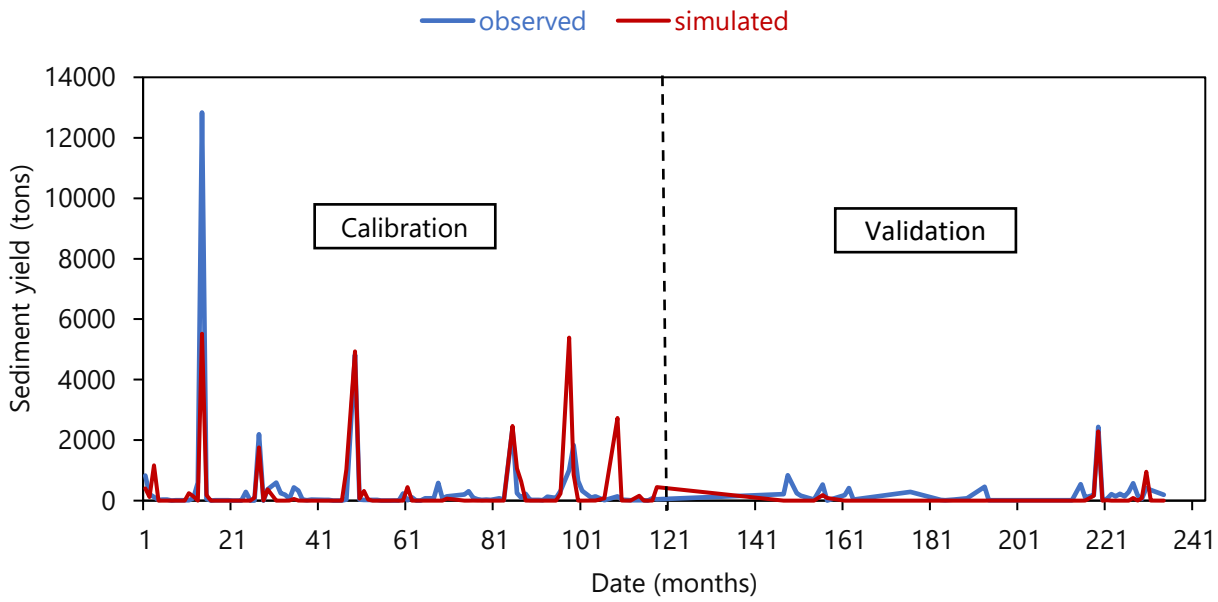


Figure 4.2. The observed and simulated monthly sediment load (a) calibration (1998-2007) and Validation (2008-2017) periods.

The high coefficient of determination (R^2) and Nash-Sutcliffe (NS) of the calibration and validation period indicate that the auto-calibrated model adequately captured the streamflow and sediment yield Sub-basin. Both the indices' values are more than 0.5, which indicates a good model performance. Statistical performance indices used in this study are shown in Table *4.3* and Table *4.4*. The model overestimates streamflow in some catchments indicated by negative Percent Bias (PBIAS) values. In some catchments, the model overestimates the observed streamflow by more than 50% (i.e., B7H014 and B1H019), which is unsatisfactory as it is way more than 25%. Such finding is attributed to the calculation of average monthly streamflow where some daily values are missing, leading to miscalculations for specific months. The root means square error (RSR) values are all less than 0.7 in all the calibrated catchments for both the calibration and validation period, indicating good model performance.

The high positive values in catchment B12C indicates that the model had a bias towards underestimating sediment yield. Such findings may result from the type of objective function used in running the calibration and validation program, which affects the given solution (Abbaspour, 2013). Therefore, in avoiding such a problem, one would have only to consider the objective function used in the calibration and validation process for model performance analysis. Overall, the results indicate that in most catchments, the model performed better during calibration than validation as in other studies (Moriasi *et al.*, 2007; Mwangi *et al* 2015; Gyamfi *et al.*, 2016A). This is attributed to the calibration and

validation process. During calibration, selected parameters are adjusted in the model iteratively to obtain the best fit between the model calculations and the monthly average observed data. Therefore, the calibration procedure ensures that the model performs better. Whereas, the validation process ensures confidence in the calibration parameters (Abbaspour *et al*., 2017).

However, the opposite is true in catchment B20B (B2H007) where the model performs better during the validation period than the calibration period. Also, in the case of B6H003, the model gives the same result (R^2 = 0.80) for both the calibration and validation period. The results also indicate that the model performs better during validation in terms of R^2 in the case of sediment load (B12C) as shown in *Table 4.3*. It is important to note that both the calibration and validation period was ten years. Some stations had more missing data during the calibration period (1998 – 2007) than the validation (2008 - 2017) which might contribute to high values of statistical performance indices during validation. Model performance during calibration and validation processes is also affected by the type and availability of data used. Data used for calibration and validation periods differ from each other because of availability, measurements conditions, techniques and constituent type (Moriasi *et al*., 2007). Model performance in the validation period can also be better than the calibration period especially if the calibration period is drier than the validation. The trend analysis conducted on the climate stations used in this study indicated a drying trend. Some possible errors and uncertainties may be responsible for some of the variability in the results across the different stations.

Table 4.3. Monthly time step calibration and validation performance indices for streamflow of outlets.

Indices	Process	B7H014	B7H013	B7H004	B6H003	B4H005	B2H008	B2H007	B1H019
R^2	Calibration	0.76	0.95	0.55	0.80	0.66	0.81	0.61	0.88
	Validation	0.64	0.68	0.60	0.80	0.56	0.55	0.84	0.74
NS	Calibration	0.76	0.93	0.54	0.76	0.63	0.80	0.54	0.81
	Validation	0.58	0.67	0.56	0.62	0.54	0.54	0.66	0.61
PBIAS	Calibration	-8.40	-0.10	3.40	-6.30	19.80	23.70	22.10	-52.40
	Validation	-147.10	-2.20	13.40	-20.00	14.70	5.20	18.00	-59.10
RSR	Calibration	0.49	0.26	0.67	0.49	0.61	0.45	0.68	0.44
	Validation	0.65	0.57	0.66	0.61	0.68	0.67	0.58	0.68

Table 4.4. Monthly time step calibration and validation performance indices for sediment yield.

Indices	Period	B12C
R^2	Calibration	**0.56**
	Validation	0.72
NS	Calibration	**0.55**
	Validation	0.54
PBIAS	Calibration	**11.10**
	Validation	61.30
RSR	Calibration	**0.67**
	Validation	0.68

Hydrological modelling comes with inherent uncertainties. The SWAT model uncertainties come from simplifying complex processes such as using the SCS curve number method for flow partitioning, unaccounted processes, lack of exact details regarding the occurrence of such process data limitation, and input data quality (Abbaspour *et al*., 2015). Such uncertainties are expected in this study since the Olifants Sub-basin is large. The uncertainties are visible in this study's prediction errors, such as the high percentage bias in B7H014 and B1H019. In this study, outlets with significant missing observed data showed considerable uncertainties.

4.4. SLM impacts on water flow and sediment yield

4.4.1. Filter Strips

Filter Strips had no impact on streamflow however, a decreased in sediment yield was reported. Because filter strips slow runoff velocity and filter out suspended sediments and associated pollutants. In contrast to other studies (Arabi et al., 2008; Mwangi et al., 2015), this study indicates a linear relationship between filter strip width and sediment yield reduction. A study by Gathagu et al. (2018) reported similar findings where an increase in filter strip width resulted in a decrease in sediment yield. Other factors contribute to sediment trapping efficiencies such as slope, vegetation, inflow rate, and particle size (Mwangi et al., 2015). The Olifants sub-basin is characterised by pasture and agricultural activities, and the slope gradient ranges from 0 to 30%, with the majority (95%) of the area falling within the slope class 0-8 per cent. These attributes contribute to the linearity of the relationship between the filter strip width and sediment yield, as shown below. Filter strips serve as a pre-treatment of runoff where pollutants and sediments are trapped providing a great service to other best management practices.

Table 4.5. The sensitivity of fitted filter strips width and their impacts on trapping sediments.

FILTER Width (m)		Sensitivity		Impact
Range (m)	Fitted (m)	t-stat	p-value	Sediment yield (%)
0-5	5	11.50	0.00	24.53
5-10	7	-1.61	0.11	37.54
10-15	10	-103.66	0.00	49.91
15-20	15	-322.36	0.00	66.96
20-25	20	-729.56	0.00	80.37
25-30	25	-84.76	0.00	91.59

The optimum filter strip width in the Olifants Sub-basin is 25 meters, with an almost 100% trapping ratio. Mainly due to the flat slope gradient of the catchment where implemented. Favourably 30-meter filter strips should be implemented in steep slope areas for maximum sediment trapping efficiency. However, due to construction cost (limited data on the actual cost) and land availability, it is difficult to recommend a filter strip of such magnitude. A filter strip of at least 5 meters as a pre-treatment service to other SLM practices would make a significant difference in the load of sediments in the sub-basin.

4.4.2. AGROFORESTRY

Total water yield (mm) and sediment yield (tons/km^2) represent the agroforestry's hydrological impacts for the agroforestry section. In three of the calibrated catchments, agricultural HRUs were converted into woodlots and parameters associated with a change in tree cover were adjusted. The area under agriculture converted into woodlots in B42F, B20B, and B12C catchments were 66%, 63%, and 45% of the total catchment area. The results indicate a decrease in total water yield and sediment yield with the introduction of agroforestry. The current study's results are similar to those reported from previous studies in terms of a decrease in net water yield in the face of an increase in tree cover (Brown *et al*., 2013; Suarez *et al*., 2014; Mwangi *et al*., 2016). The decrease in water yield is compensated by an increase in actual evapotranspiration (*Table 4.6*). Areas under tree cover have high evapotranspiration rates as they remain wet much of the year; thus, evaporation rates can be twice as much compared to other land covers.

Surface runoff decreased by approximately 52%, 14% and 16% in catchment B42F, B20B and B12 C when the HRUs under agriculture in the catchments were converted to agroforestry systems. The decrease in surface runoff is attributed to an increase in

percolation (*Table 4.6*). Trees in agroforestry systems increase infiltrability and preferential flow, thus decreasing surface runoff and increasing percolation (Benegas *et al.*, 2014; Qazi *et al.*, 2017). Increasing tree cover in the former agricultural fields results in improved hydrological conditions where more rainwater gets absorbed into the soil and surface runoff is reduced (Mwangi *et al.*, 2016). Improved infiltration may not occur immediately after the agroforestry implementation as it takes time for catchments to make significant hydrological changes (Brown *et al.*, 2013).

Sediments are mainly transported through surface runoff, so it is plausible to record a decrease in sediment yield consequently. Additionally, agroforestry increases land productivity and protects soil from erosion, contributing to a decrease in sediment yield. The impacts of precipitation and surface runoff usually become negligence in sediment yield in agroforestry systems due to improved soil conditions and vegetation characteristics (Zhu *et al.*, 2018). Multilayers of canopy and litter associated with agroforestry contribute to reduced runoff and sediment yield (Zhu *et al.*, 2018). The understory vegetation cover in agroforestry systems plays a vital role in reducing the erosive power of rainfall by increasing surface roughness and reducing surface runoff, which translates to reduced sediment yield (Liu *et al.*, 2017). A reduction in sediment yield is also attributed to increased leaf area, canopy cover, stand litter, and leaf area index (Zhu *et al.*, 2018). A study by Kinnell (2010) indicates that small raindrops may gather on the leaves and branches and become more giant raindrops, intensifying the power of raindrops and increasing soil erosion. Some studies also reported an increase in sediment yield from forested catchments due to soil type and underlying geology (Verbist *et al.*, 2010). Various factors influence processes that occur in nature; for these reasons, findings from studies differ.

Table 4.6. Changes in hydrological processes with the introduction of agroforestry.

Variables	B42F		B20C		B12C	
	Baseline	Agroforestry	Baseline	Agroforestry	Baseline	Agroforestry
Water yield (mm)	98.40	74.76	13.81	10.64	73.23	17.23
Surface runoff (mm)	49.26	15.70	13.31	10.13	41.74	17.12
Percolation (mm)	151.40	182.37	65.30	68.11	50.25	69.68
Evapotranspiration (mm)	508.56	511.20	636.17	636.46	529.87	534.64
Sediment Yield (t/km^2)	12.10	4.00	1.20	0.80	2.70	0.30

Plant water use or uptake also contributes to a decrease in the total water yield of a catchment. Therefore, it is crucial to take into account the species of plants implemented in agroforestry systems. Agroforestry systems are associated with high water use. It is essential to consider the natural climatic issues of water availability in the catchment, selection of tree species and forest management options (Ferraz *et al.*, 2013) when introducing agroforestry systems. Implementation of indigenous species or varieties in the catchment results in low hydrological impacts and plays a vital role in reducing and regulating water use compared to introduced tree species (Everson *et al.*, 2011; Ferraz *et al.*, 2013). Agroforestry efforts should adopt management strategies that address the shortage in the water supply (water availability) by regulating the area under tree cover in a catchment (Garcia *et al.*, 2018).

4.4.3. CONTOUR FARMING

Simulation of contour farming by changing the parameter CN2 and USLE_P values based on a slope percentage range of the HRU leads to decreased surface runoff, streamflow and sediment yield. Streamflow and sediment yield decreased by approximately 34% (19.42 to 9.65 m^3/s) and 46% (10514.15 to 3935.41 tons), respectively (). The current study's findings are similar to those of Mwangi *et al.* (2015) and Gathagu *et al.* (2018), who also found a decrease in surface runoff and sediment yield due to contour farming. Sediment yield decreased by 49% in the Sasumua watershed and 36% in the Thika-Chania catchment, both found in Kenya.

Table 4.7. The impacts of contour farming on streamflow and sediment yield.

Variable	Baseline	Contour Farming
Streamflow (m^3/s)	19.42	9.65
Sediment yield (tons)	10514.14	3935.41

The study results indicate that contour farming increases rainwater that infiltrates into the soil, reducing the amount of water left to surface runoff. Consequently, decreasing the number of sediments and water flowing in streams. Percolated rainwater may re-enter streamflow as return flow, particularly during dry seasons. Some studies indicate contour farming is also more effective when combined with other SLM practices such as grassed waterways and filter strips (Mwangi *et al.*, 2015; Gathagu *et al.*, 2018). Contour farming is one of the most cost-effective and less labour-intensive SLM practices.

4.4.4. PARALLEL TERRACES

Simulation of parallel terraces in the Sub-basin reduced sediment yield by 87% from an average annual of 10514.14 tons to 759.17 tons showing its potential as an SLM measure (). Streamflow was reduced by only 20% from an average annual baseline of 19.42 m^3/s to 12.99 m^3/s, mainly due to a decrease in surface runoff (). The significant decrease in sediment is attributed to the adjustment of slope length (SLSUBBSN) and USLE practice factor (USLE_P) parameters. These parameters are sensitive to sediment yield, particularly the USLE practice factor. USLE_P parameter is sensitive to sediment yield with a decrease in value resulting in sediment yield reduction (Ullrich and Volk 2009).

Table 4.8. The impacts of parallel terraces on streamflow and sediment yield.

Variable	Baseline	Parallel Terraces
Streamflow (m^3/s)	19.42	12.99
Sediment yield (tons)	10514.14	759.17

Other studies also found similar results where there was a decrease in sediment yield and surface runoff due to parallel terraces (Arabi *et al.*, 2008; Mwangi *et al.*, 2015). The reduction in surface runoff and sediment yield comes about because parallel terraces reduce slope length and peak runoff rate (Arabi *et al.*, 2008). It is important to note that the impact of parallel terracing on sediment yield is more significant than water yield as they reduce more sediments per unit area (Mwangi *et al.*, 2015; Bai *et al.*, 2019).

4.5. Summary

This study provides a predictive method to quantitatively assess the impact of SLM practices and technologies on runoff and sediment yield at the Sub-basin scale. Calibration and validation results for SWAT show that the adopted approach provides relatively good results, allowing its use in scenario analysis. SLM practices, namely filter strips, agroforestry, contour farming and parallel terraces, positively impact surface runoff and sediment yield. However, these practices should be implemented based on local biophysical and socio-economic characteristics, and care should be taken to ensure sustainable water availability. The results have significant implications for all stakeholders involved in water and land resources management and at all levels. The findings of this study are especially useful for land users, watershed managers and policy decision-makers who are tasked with implementing and maintaining management projects. Advancing and

sharing knowledge surfaced in this study may encourage land users within the study area to adopt SLM practices and technologies. Watershed managers or water management area authorities can use the findings to devise informed and integrated management plans for the Olifants Sub-basin. The results can also be a useful tool for policymakers to provide an enabling environment for the adoption and upscaling of SLM practices and technologies in the Olifants Sub-basin.

5. CONCLUSION AND RECOMMENDATIONS

In this study, the hydrological impacts of selected Sustainable Land Management (SLM) practices and technologies were assessed using the SWAT model. The assessment of the impacts of implementing SLM measures is a cost-effective way to provide decision-makers with information such as the effectiveness of specific practices before implementation. Such an approach potentially leads to a higher success rate. With regards to contour farming, filter strips, parallel terraces and agroforestry impacts on streamflow and sediment yield, corresponding parameters were modified accordingly in the SWAT model. The results indicate that the SWAT model was able to satisfactorily simulate the hydrological processes of the Olifants Sub-basin. Contour farming, filter strips, parallel terraces significantly improved streamflow and decreased sediment yield in the study area. On the hydrological impacts of agroforestry, the results show that agroforestry significantly decreases sediment yield and total water yield. The decrease in total water yield by agroforestry is attributed to increased tree cover and evapotranspiration.

When adopting agroforestry to address land degradation it is important to consider plant species that are indigenous to the area and management strategies that address their impacts on the water resources. A filter strip with a 25 meters width is recommended in the Sub-basin. It is important to note that this recommendation is solely based on empirical evidence that did not account for construction cost. Contour farming should be combined with other practices such as filter strips to increase its effectiveness. As for parallel terraces, they should be implemented in the agricultural field with high erosion rates to halt sediment yield.

Ongoing data collection and further research on the impacts of Sustainable Land Management (SLM) after implementations in the Olifants Sub-basin should be conducted for further validation. The lack of maintenance of some of the datasets from the DWS is a serious cause for concern. Observation data of good quality and integrity are a crucial requirement of the SWAT model. Therefore, DWS should not only attempt to provide the data but also ensure that they as adequately maintained, verified, and patched. SWAT users should also take it upon themselves to verify and patch missing data values using trusted methods to increase their models' credibility and performance.

Sediment load data in tons is not available in the Sub-basin. Only Total Suspended Solids (TSS) data are available and can be used to estimate sediment load. The estimation of

sediment load provides a reasonable basis for sediment calibration. The sediment load is estimated from TSS and streamflow. The TSS (Mg/L) is multiplied by the flow of that particular day when the sample was taken. There is a need for an aggressive campaign to collect primary flow and sediment data to improve the plausibility of results.

The methods used to represent SLM in the present study are based on historical and ongoing experimental trials of conservation practices. They may not capture the unique processes that occur specifically in Olifants Sub-basin. Despite their limitations, the results can be used to guide the development of watershed management programs and to evaluate existing management strategies. Filter strips, agroforestry, contour farming and parallel terraces can be essential tools in arresting land degradation in the Sub-basin. These practices have positive impacts on surface runoff and sediment yield. A socioeconomic survey and benefit-cost analysis should also be carried out to determine farmers' preferences and each intervention option's cost-effectiveness.

This predictive modelling study can be used as a decision support tool for implementing SLM practices to address land degradation. The SWAT model heavily relies on input data. Therefore, sufficient and accurate data is required to calibrate and validate streamflow and sediment yield for the results to be reliable. Calibration and validation results from the SWAT model indicate that the adopted approach provides relatively good results, allowing its use in risk analysis.

For further studies, it is important to patch and improve hydroclimatic data quality using statistical and data science tools. This will contribute to the overall quality of the SWAT model. Future studies need to employ computer-based programs such as Loadset to predict sediment load based on TSS and discharge to limit errors. Proper estimation of streamflow and sediment yield generation is essential for optimal modelling and water resources management. It is recommended that future research include an investigation of the effects of climate change on the effectiveness of SLM technologies and practices at the catchment scale. Such an investigation could provide valuable information on the differences in SLM effectiveness at catchment scale in varying climate conditions. Future watershed modelling studies should also consider the sensitivity and uncertainty of SLM practices and technology to changes in climate to help watershed managers and policy decision-makers understand the risk and reliability of implementing SLM measures and maintaining SLM investments in a changing climate.

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APPENDICES

Appendix A

Example of the land types properties

LAND TYPE / *LANDTIPE* : **Ab29**

CLIMATE ZONE / *KLIMAATSONE* : 11418

Area / *Oppervlakte* : 5339 ha

Estimated area unavailable for agriculture

Beraamde oppervlakte onbeskikbaar vir landbou : 20 ha

Occurrence (maps) and areas / *Voorkoms (kaarte) en oppervlakte* :

2530 Barberton (5339 ha)

Inventory by / *Inventaris deur* :

J L Schoeman

Modal Profiles / *Modale profiele* :

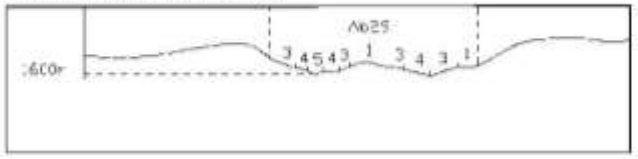
None / Geen

Terrain unit / *Terreineenheid*	1	3	4	5
% of land type / *% van landtipe*	10	45	40	5
Area / *Oppervlakte (ha)*	534	2403	2136	267
Slope / *Helling (%)*	5 - 8	5 - 15	2 - 3	0 - 2
Slope length / *Hellingslengte (m)*	100 - 300	200 - 500	100 - 300	50 - 200
Slope shape / *Hellingsvorm*	Y	Y	X	X
MB0, MB1 (ha)	214	1682	2136	267
MB2 - MB4 (ha)	320	721	0	0

Soil series or land classes / ***Grondseries of landklasse***	**Depth** / ***Diepte*** (mm)	**MB**	1 ha	1 %	3 ha	3 %	4 ha	4 %	5 ha	5 %	**Total** / ***Totaal*** ha	**Total** / ***Totaal*** %	**Clay content %** / ***Klei-inhoud %*** A	E	B21	Hor	**Texture** / ***Tekstuur*** Class / *Klas*	**Depth limiting material** / ***Diepte-beperkende materiaal***
Soil-rock complex																		
Grond-rotskompleks:																		
Rock/Rots		4	107	20	240	10					347	6.5						
Farningham Hu17, Doveton Hu27, Argent Sd11	200-400	2	214	40	481	20					694	13.0	30-45		35-60	B	SaCl,ClLm,Cl	so,R
Balmoral Hu18, Vimy Hu28	600-1200+	0	53	10	721	30	1068	50			1842	34.5	35-50		55-70	B	Cl	so
Farningham Hu17, Doveton Hu27	600-1200+	0	80	15	481	20	641	30			1201	22.5	35-45		40-55	B	SaCl,ClLm,Cl	so
Argent Sd11, Richmond Sd12	600-1200+	0	80	15	481	20	214	10			774	14.5	35-45		40-70	B	Cl	so
Stanger Bo11, Glengazi Bo31	600-1200+	0					214	10	80	30	294	5.5	35-50		45-60	A	SaCl,ClLm,Cl	so
Rydalvale Ar30, Arcadia Ar40	700-1200+	0							187	70	187	3.5	40-60			A	Cl	so

Terrain type / *Terreintipe* : A3

Terrain form sketch / *Terreinvormskets*



For an explanation of this table consult LAND TYPE INVENTORY (table of contents)

Ter verduideliking van hierdie tabel lyk LANDTIPE - INVENTARIS (inhoudsopgawe)

Geology: Gabbro and norite of the Rustenburg Layered Suite, Bushveld Complex.

Geologie: Gabbro en noriet van die Gelaagde Suite Rustenburg, Kompleks Bosveld.

Appendix B

HRUs Land use, Soil and Slope distribution report

SWAT model simulation Date: 6/9/2020 12:00:00 AM Time: 00:00:00
MULTIPLE HRUs Land Use/Soil/Slope OPTION THRESHOLDS: 10 / 10 / 10 [%]
Number of HRUs: 1246
Number of Subbasins: 124

	Area [ha]	Area[acres]
Watershed	5397615.5136	13337777.8149

LANDUSE:	Area [ha]	Area[acres]	%Wat.Area
Range-Brush --> RNGB	2962069.7610	7319422.4830	54.88
Residential-Med/Low Density --> URML	121296.3304	299729.2972	2.25
Pasture --> PAST	1273397.0986	3146627.9005	23.59
Agricultural Land-Row Crops --> AGRR	90067.4718	22561.2262	1.67
Southwestern US (Arid) Range --> SWRN	803.0505	397.5279	0.07
Forest-Evergreen --> FRSE	38337.3239	94733.4443	0.71
Agricultural Land-Generic --> AGRL	876785.9552	2166581.9347	16.24
Orchard --> ORCD	9864.6080	24375.9395	0.18
Water --> WATR	871.4366	2153.3633	0.02
Industrial --> UIDU	21122.4776	52194.6983	0.39
SOILS:			
Ae126	22484.0961	55559.3257	0.42
Ea165	31520.8447	77889.5832	0.58
Ae125	7255.9206	17929.7427	0.13
Ae127	7271.8140	17969.0159	0.13
Bd33	9647.0617	23838.3718	0.18
Fa350	5651.3041	13964.6551	0.10
Ab53	1129.1124	2790.0932	0.02
Fb179	125527.1652	310183.9016	2.33
Ae128	46156.5846	114055.2284	0.86
Fb180	93733.5455	231620.2776	1.74
Ea89	50996.8021	126015.6477	0.94
Fb183	120617.5346	298051.9590	2.23
Fb181	167422.7965	413710.1014	3.10
Ea90	1145.2505	2829.9713	0.02
Ea91	456.1733	1127.2271	0.01
Fb184	88158.5048	217844.0732	1.63

Fb186	12635.1753	31222.1500	0.23
Fb175	29992.4794	74112.9163	0.56
Fb176	27303.6120	67468.5905	0.51
Fb177	45792.1887	113154.7878	0.85
Fa347	16514.1782	40807.3600	0.31
Ib157	67446.4801	166663.6246	1.25
Ib180	6536.6694	16152.4370	0.12
Ib181	39853.3746	98479.6813	0.74
Ab52	1989.9649	4917.3028	0.04
Ae175	28249.2879	69805.4028	0.52
Ah114	10285.4868	25415.9521	0.19
Ib155	188583.4664	465999.1746	3.49
Ib302	23656.5833	58456.6002	0.44
Ab89	3197.9425	7902.2758	0.06
Fa538	1607.8421	3973.0582	0.03
Fb171	42134.9223	104117.4997	0.78
Ab55	4650.5366	11491.7084	0.09
Ib182	1903.4459	4703.5101	0.04
Fb178	36449.0918	90067.5283	0.68
Ib185	16317.8486	40322.2198	0.30
Ib453	45195.5438	111680.4485	0.84
Ae352	8793.4564	21729.0704	0.16
Ah81	21144.7877	52249.8275	0.39
Ia176	11601.7561	28668.5194	0.21
Ae348	2050.4036	5066.6499	0.04
Fb182	14039.4449	34692.1704	0.26
Ae339	24631.7749	60866.3473	0.46
Db244	665.9455	1645.5847	0.01
Ia174	10651.4597	26320.2895	0.20
Ae343	16460.6447	40675.0760	0.30
Ae346	6739.1809	16652.8530	0.12
Ia175	2123.8146	5248.0521	0.04
Ae115	25459.1864	62910.9226	0.47
Ea88	46324.4973	114470.1490	0.86
Ib190	11697.5473	28905.2243	0.22
Ic154	37773.7054	93340.7148	0.70
Ae122	6423.8910	15873.7558	0.12
Ib454	1197.5320	2959.1614	0.02
Ab40	2451.0349	6056.6298	0.05
Ab41	2837.2514	7010.9900	0.05
Ab59	1702.3082	4206.4887	0.03
Ae341	17176.7789	42444.6795	0.32
Ae354	21076.9751	52082.2594	0.39

Ae233	29444.3070	72758.3548	0.55
Fa279	37085.4165	91639.9184	0.69
Ib293	22672.3413	56024.4891	0.42
Ae340	3646.8658	9011.5876	0.07
Ae356	3091.8122	7640.0226	0.06
Ae387	1266.7798	3130.2763	0.02
Fc476	42714.7523	105550.2886	0.79
Ae353	1344.5653	3322.4882	0.02
Ae230	7238.3994	17886.4468	0.13
Ea148	29990.8110	74108.7934	0.56
Hb1	12102.8519	29906.7522	0.22
Ac154	17071.9711	42185.6942	0.32
Ae89	28970.5718	71587.7315	0.54
Ea145	29017.3396	71703.2971	0.54
Ae351	19403.5209	47947.0703	0.36
Ae130	10042.5697	24815.6920	0.19
Fa359	5236.0539	12938.5509	0.10
Ic157	4663.5151	11523.7790	0.09
Ae123	29419.4882	72697.0264	0.55
Fb172	8255.1637	20398.9222	0.15
Ae342	3312.4189	8185.1527	0.06
Ae386	3043.7951	7521.3699	0.06
Fa351	21485.9308	53092.8093	0.40
Fa352	55833.9919	137968.5857	1.03
Ae101	32912.1849	81327.6546	0.61
Ae228	19989.2251	49394.3747	0.37
Fa764	60133.0262	148591.7145	1.11
Ac89	14889.3524	36792.3342	0.28
Fa358	4621.4181	11419.7551	0.09
Ib193	6855.7442	16940.8868	0.13
Ib194	3407.5912	8420.3283	0.06
Ic158	4928.3304	12178.1508	0.09
Ah78	8398.5673	20753.2796	0.16
Fa283	9397.5301	23221.7668	0.17
Ac150	22096.4707	54601.4839	0.41
Bb89	14780.7543	36523.9829	0.27
Fc477	11801.0216	29160.9144	0.22
Ae350	14557.6669	35972.7228	0.27
Ba63	70095.9582	173210.6176	1.30
Ib452	10562.2547	26099.8594	0.20
Ae349	6172.9427	15253.6501	0.11
Ae338	3571.3901	8825.0836	0.07
Ah77	9627.0741	23788.9813	0.18

Ae107	14888.9937	36791.4479	0.28
Ae119	7130.6677	17620.2365	0.13
Ib191	10143.1219	25064.1613	0.19
Ib197	11149.4476	27550.8426	0.21
Bc57	1378.7075	3406.8552	0.03
Bc9	20589.5732	50877.8649	0.38
Fa763	17412.5440	43027.2669	0.32
Ae27	24384.8829	60256.2649	0.45
Ib192	15242.2247	37664.2993	0.28
Ae120	1886.3859	4661.3538	0.03
Fa354	13096.3048	32361.6240	0.24
Fa355	15905.7941	39304.0126	0.29
Ab30	4940.7107	12208.7432	0.09
Ab32	8865.5965	21907.3322	0.16
Ab57	3811.6183	9418.6994	0.07
Ae226	26658.4906	65874.4632	0.49
Ae23	36994.9408	91416.3485	0.69
Bb90	13107.2452	32388.6582	0.24
Ae231	16709.7974	41290.7449	0.31
Ea1	90832.3110	224451.1821	1.68
Ib24	12117.3743	29942.6377	0.22
Bd4	149321.1106	368979.9303	2.77
Ib300	2918.2582	7211.1620	0.05
Dc31	20234.8068	50001.2193	0.37
Ib30	38371.7643	94818.5483	0.71
Bc7	10745.5933	26552.8984	0.20
Fb534	918.1972	2268.9112	0.02
Fa327	17207.6585	42520.9846	0.32
Ib156	2857.4130	7060.8105	0.05
Ac75	10036.0566	24799.5976	0.19
Ac76	2565.6193	6339.7736	0.05
Fa328	2155.7488	5326.9631	0.04
Fa343	31453.0024	77721.9415	0.58
Ib154	15922.2945	39344.7859	0.29
Ib31	18003.2013	44486.8105	0.33
Ab29	2918.8801	7212.6987	0.05
Ib32	12162.7677	30054.8071	0.23
Bc4	7774.6571	19211.5665	0.14
Ib19	41226.0197	101871.5560	0.76
Ae18	11729.1238	28983.2513	0.22
Ae19	37592.0955	92891.9477	0.70
Bd2	2942.8777	7271.9980	0.05
Fa6	5132.7127	12683.1896	0.10

Ae26	12854.3678	31763.7855	0.24
Ib23	1394.4524	3445.7615	0.03
Ae20	39918.4815	98640.5638	0.74
Ac151	27039.0643	66814.8798	0.50
Ae24	2703.1427	6679.6009	0.05
Bc2	12641.4850	31237.7416	0.23
Bc3	5726.1153	14149.5171	0.11
Bd3	13619.7739	33655.1424	0.25
Dc1	26526.4776	65548.2526	0.49
Ea7	5086.6555	12569.3802	0.09
Ib33	20984.8199	51854.5393	0.39
Ab3	9793.0776	24199.1844	0.18
Ea5	65275.2123	161298.3133	1.21
Ib20	1817.1860	4490.3574	0.03
Ac74	4493.2167	11102.9632	0.08
Ba66	35207.0611	86998.4082	0.65
Fb170	21412.8118	52912.1287	0.40
Ba65	22106.7415	54626.8635	0.41
Fa325	13158.8762	32516.2410	0.24
Ib11	15378.8398	38001.8821	0.28
Ib12	6823.1235	16860.2793	0.13
Ib8	2419.8180	5979.4913	0.04
Ea6	31916.4996	78867.2663	0.59
Ib27	9007.8115	22258.7526	0.17
Ba16	2649.5582	6547.1907	0.05
Ac1	9418.6278	23273.9003	0.17
Ac2	37711.3932	93186.7382	0.70
Ea4	10659.5327	26340.2383	0.20
Fb3	10472.3497	25877.6996	0.19
Ib21	4829.9607	11935.0744	0.09
Ib22	23412.9510	57854.5725	0.43
Bc1	17302.2550	42754.7372	0.32
Ib10	44747.1597	110572.4690	0.83
Ba11	69967.8523	172894.0615	1.30
Fa7	45642.8039	112785.6506	0.85
Ae25	5864.7658	14492.1296	0.11
Ib13	6038.5661	14921.5987	0.11
Ba15	3635.6394	8983.8467	0.07
Bb16	28878.1673	71359.3954	0.54
Ib15	29404.5117	72660.0186	0.54
Ib16	23919.3328	59105.8672	0.44
Ba37	91019.9505	224914.8486	1.69
Ib9	34215.7151	84548.7427	0.63

Bb9	16285.7259	40242.8431	0.30
Fa5	5843.0188	14438.3915	0.11
Ba13	43829.9985	108306.1177	0.81
Bb12	60588.0437	149716.0854	1.12
Bb11	21089.6407	52113.5568	0.39
Ib29	6725.2819	16618.5077	0.12
Ba14	20746.2588	51265.0428	0.38
Ba18	19496.0178	48175.6349	0.36
Ba19	86481.2111	213699.3966	1.60
Bb14	31281.3667	77297.8212	0.58
Ib28	11743.6813	29019.2236	0.22
Bb8	8110.6987	20041.9420	0.15
Ba12	7971.5055	19697.9887	0.15
Ba17	1230.9887	3041.8347	0.02
Ib25	4775.3503	11800.1293	0.09
Ib34	4983.1644	12313.6485	0.09
Ib35	15929.4637	39362.5014	0.30
Ba20	4963.3448	12264.6731	0.09
Ea8	11449.9965	28293.5138	0.21
Bb10	21544.4790	53237.4849	0.40
Ib17	3269.3101	8078.6288	0.06
Bb17	28184.5186	69645.3548	0.52
Bb13	41488.8581	102521.0427	0.77
Ba4	69764.3399	172391.1722	1.29
Ba5	107099.9757	264649.3950	1.98
Bb4	237061.1260	585789.8954	4.39
Fa8	14547.4726	35947.5323	0.27
Ba6	13512.2720	33389.4997	0.25
Ab5	6632.9024	16390.2335	0.12
Ba3	25652.6016	63388.8612	0.48
Ba2	15972.1258	39467.9215	0.30
Bb3	49431.6030	122147.9625	0.92
Bb5	25190.1442	62246.1059	0.47
Ba22	53223.1093	131516.9641	0.99
Ab9	25817.0917	63795.3245	0.48
Ea20	68335.0777	168859.3936	1.27
Ba33	3116.4005	7700.7814	0.06
Ea23	8066.8063	19933.4818	0.15

SLOPE:

0-8	3883256.4484	9595720.8469	71.94
8-30	1056376.8503	2610360.0159	19.57
30-9999	457982.2149	1131696.9521	8.48