Institute of Water and Energy Sciences (Including Climate Change)

IMPACT EVALUATION OF SOIL AND WATER CONSERVATION MEASURES ON ECOSYSTEM SERVICES IN THIKA-CHANIA CATCHMENT, KENYA

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Declaration and Recommendation

I **John Ng'ang'a Gathagu**, hereby declare that this thesis represents my personal work, realized to the best of my knowledge. I also declare that all information, material and results from other works presented here, have been fully cited and referenced in accordance with the academic rules and ethics.

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This thesis is the candidate's original work and has been prepared with our guidance and assistance and therefore recommended for examination with our approval as the official university supervisors.

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Abstract

Across the world, ecosystem services such as water provision, sediment trapping and floods regulation are threatened by the increased degradation of catchments. This is as a result of increased pressure from population growth, improper farming methods and encroachment into riparian areas leading to soil erosion and water quality deterioration. These ecosystem services could be restored by implementing best management practices to enhance flow regulation and control sediment yield. In Thika-Chania catchment, the failing services are manifested in the deterioration of water quality, sedimentation of reservoirs and inadequate surface water in dry seasons. The aim of this study was to evaluate the impacts of implementing soil and water conservation measures on water and sediment yield. The Soil Water and Assessment Tool (SWAT) was used to evaluate contour farming, filter strips, terraces and grassed waterways on water and sediment yield. Calibration and validation of SWAT to simulate streamflow was conducted using SWATCUP-SUFI2 at RGS 4CB05. Results indicated respective monthly NSE, coefficient of determination and PBIAS of 0.66, 0.69 and 10.3 and 0.73, 0.75 and 7.2 during calibration and validation. Sediments calibration and validation was achieved using data obtained from a bathymetric survey conducted in the catchment. The average annual sediment yield was found to be 21.507 t/ha at the outlet of the catchment and average annual surface runoff of 202.28 mm. Model parameters were adjusted to simulate the impacts of contour farming, filter strips, terraces and grassed waterways. Terraces were found to be most effective in reducing sediment yield by 80.70% while grassed waterways, 3m-filter strips and contour farming reduced average annual sediment yield by 53.97%, 46.04% and 35.81%. A combination of various soil and water conservation methods were also evaluated. Surface runoff and total water yield was only influenced by contour farming and terraces. We concluded that soil and water conservation methods are effective in improving the provisioning and regulating services of the ecosystem. Combining more than one conservation method would have a greater impact on sediment yield reduction. A cost benefit analysis of the conservation methods need to be carried out before their implementation at the catchment or sub-catchment level.

Keywords: Calibration, Validation, Terraces, Contour farming, filter strips, grassed waterways, RGS

Résumé

Dans le monde entier, les services écosystémiques tels que l'approvisionnement en eau, la purification, le piégeage des sédiments et la régulation des inondations sont menacés par la dégradation accrue des bassins versants résultant de la pression accrue de la croissance démographique, des méthodes agricoles inappropriées et de l'empiètement dans les zones riveraines. Ces services écosystémiques pourraient être restaurés en mettant en œuvre les meilleures pratiques de gestion afin d'améliorer la régulation des flux et de contrôler le rendement des sédiments. Dans le bassin versant de Thika-Chania (840 km^2), les services défaillants se manifestent par la détérioration de la qualité de l'eau et la sédimentation des réservoirs ayant pour conséquence l'augmentation des coûts de traitement de l'eau en raison de sa forte turbidité. L'objectif de cette étude était d'évaluer les impacts des mesures de conservation des sols et de l'eau sur les rendements de gestion des écoulements et de contrôle des sédiments qui sont des éléments jouant un rôle essentiel dans l’équilibre des écosystèmes. L'outil SWAT a été utilisés pour évaluer l'élevage du contour, les bandes de filtre, les terrasses et les voies d'eau gazonnées. L'étalonnage et la validation du débit ont été effectués à l'aide de SWATCUP-SUFI2 à la station de mesure 4CB05. Les résultats ont indiqué respectivement que le NSE mensuel, le coefficient de détermination et le PBIAS sont de 0,66, 0,69 et 10,3 et 0,73, 0,75 et 7,2 lors de l'étalonnage et de la validation. L'étalonnage et la validation des sédiments ont été effectués en utilisant les données obtenues à partir d'une étude bathymétrique effectuée dans le bassin versant. Le facteur p et le facteur r respectifs étaient respectivement de 0,70, 0,67 et 0,61 et 0,45 pour l'étalonnage et la validation. On a constaté que le rendement annuel moyen des sédiments était de 21.507 t/ha à la sortie du bassin versant et un écoulement de surface annuel moyen de 202.28 mm. Les paramètres du modèle ont été ajustés pour simuler les impacts de l'agriculture de contour, des bandes de filtre, des terrasses et des voies d'eau gazonnées. Les terrasses ont été les plus efficaces pour réduire le rendement des sédiments de 80,70%. Les voies d'eau gazonnées, les bandes filtrantes de 3 m et l'agriculture de contour ont réduit le rendement annuel moyen des sédiments de 53,97%, 46,04% et 35,81%. Une combinaison de voies d'eau gazonnées, de terrasses et de bandes de 3 m a réduit le rendement annuel des sédiments de 90,42%. Le contour et les bandes de filtre ont réduit les sédiments de 62,80% par rapport à la valeur de base. La réduction des sédiments de 88,72% et 70,00% a été obtenue grâce à la simulation de voies d'eau gazonnées + terrasses et voies d'eau gazonnées + culture de contour respectivement. Le ruissellement de surface et le rendement total en eau n'a été influencé que par l'agriculture de contour et les terrasses. Au total, 18 simulations ont été

effectuées. Nous concluons que les méthodes de conservation des sols et de l'eau sont efficaces pour améliorer les services d'approvisionnement et de régulation de l'écosystème. Une analyse coût-bénéfice des méthodes de conservation doit être effectuée avant leur mise en œuvre au niveau du bassin versant ou du sous-bassin versant.

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John Ng'ang'a Gathagu

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Abbreviations

ARIES	Artificial Intelligence for Ecosystem Services
BMP	Best Management Practice
CSA	Critical Source Area
DEM	Digital Elevation Model
FILTERW	Filter strip Width
GWATD	Depth of the Grassed Waterways
GWATS	Average Slope of the Grassed Waterways
HRU	Hydrologic Response Unit
InVEST	Integrated Valuation of Ecosystem Services
ISRIC	International Soil Reference and Information Center
MGT_OP	Management Operation
MIMES	Multiscale Integrated Models for Ecosystem Services
NCSWC	Nairobi City Water and Sewerage Company
OPAL	Offset Portfolio Analyzer and Locator
SCS	Soil Conservation Service
SRTM	Shuttle Radar Topography Mission
SUF	Sequential Uncertainty Fitting
SWAT	Soil and Water Assessment Tool
TERR_SL	Terrace Slope Length
USLE	Universal Soil Loss Equation
VIC	Variable Infiltration Capacity
WISE	World Inventory of Soil Emission potential
WRMA	Water Resources Management Authority

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CHAPTER 1
INTRODUCTION

1.1 Background Information

Human beings benefit from a wide range of ecosystem services provided by nature across the world. These services however, are affected by the unsustainable use of the ecosystems due to the global population growth. Across the world, soil erosion is an ecosystem problem that reduces the productivity of soil and water quality (Humberto and Rattan, 2008). Approximately 50% of the arable land in the world is impacted by soil loss leading to deterioration of ecosystem services from which people derive their livelihoods (Cohen *et al.,* 2006). These services include the provision of fresh water, sediments regulation and global carbon cycling among others.

In Africa, soil degradation with severe impacts on ecosystems that ultimately affect food security and water quality has been documented (Tolessa *et al.* 2017; Cohen *et al.,* 2006; Lufafa *et al.,* 2003;). In the Lake Victoria Basin (LVB), the highest annual soil loss has been reported in crop lands and rangelands. According to Lufafa *et al.* (2003), croplands in the LVB, which serves an important agricultural area in East Africa, has annual soil loss of 93 t/ha. Tolessa *et al. (*2017), reported that changes in forest cover for agricultural purposes in the cental highlands of Ethiopia has led to loss of ecosystem services value estimated at US$ 3.69 million. The reduced ecosystem services include the provison of raw material, cycling of nutrients and erosion reduction (Tolessa *et al.,* 2017). Approximately 5 t/ha of top soil from agricultural lands is washed to the water bodies annually in Africa (Angima *et al.,* 2003). When soil is transported in surface runoff it causes sedimentation of reservoirs and create a risk of flooding in low lying areas besides the deterioration of water quality. Although there are some soil and water conservation efforts at the local level to control and improve the ecosystem services, soil and water resources continue to decline across the continent (Wijdenes and Bryan, 2001).

Studies in Kenya have shown that due to expansion of agricultural activities in the past decades, the original vegetation cover has greatly been removed (Maeda *et al.*, 2010; Vogl *et al.*, 2017). According to Maeda *et al.* (2010), agricultural activities have resulted to only 1% of the original vegetation in Taita Hills being preserved. Maeda *et al.,* (2010) further noted that with the existing trends, agricultural areas will increase and certainly lead to increased soil erosion thus affecting majority of the ecosystem services in the region. Erosion losses in Kenya are in equal magnitude to the agricultural exports equivalent to $ 390 million annually or 3.8% of the Gross Domestic Product (GDP) (Cohen *et al.,* 2006). Agricultural areas have the highest predicted soil loss and the magnitude of the impacts varies spatially across the country (Hunink and Droogers, 2011; Wijdenes and Bryan, 2001). At least 30 million USD are used annually in Kenya due to degradation of water resources (Mogaka *et al.*, 2006). These costs translate to about 0.5 percent of the national GDP of Kenya. Soil degradation due to erosion and nutrient mining and the inadequate soil and water conservation methods have resulted to severe decline in agricultural produce in Kenya (Lufafa *et al.,* 2003). Soil and water conservation methods could be used to reduce soil erosion and improve water quality downstream thus improving the ecosystem services (Brunner *et al.,* 2008; Herweg and Ludi, 1999; Kyalo *et al.,* 2014; Parajuli *et al.,* 2008; Tuppad *et al.,* 2010).

Thika-Chania catchment is an agricultural area that serves an important role as the source of water consumed in Nairobi city and surrounding areas (Vogl *et al.*, 2017). Soil erosion in the catchment has resulted to sedimentation and subsequent loss of water storage volume in downstream reservoirs (Archer, 1996; Hunink and Droogers, 2015). While the main source of livelihood in the catchment is agriculture, (Vogl *et al.*, 2017), a baseline survey carried out in the catchment reported that 77% of the respondent have soil erosion occurring in their farms and 54% of the inhabitants have less than 25% of their land under soil conservation methods (Leisher, 2013). Approximately 53% of the respondent noted a declining vegetation cover on their lands while at the same time 79% observed a greater deterioration of water quality in rain seasons compared to half a decade ago. This implies that soil erosion continues to increase as vegetation cover on the land decreases. Hunink *et al* (2012) highlighted the need to implement soil and water conservation measures in the catchment to reduce soil loss, improve crop yields and water quality. However, the effectiveness of each conservation method in reducing soil loss and water yield need to be established to ensure optimal benefits and save on costs of implementation.

Different soil and water conservation methods have been implemented across Africa. They include contour farming, grassed waterways, terraces, mulching, strip cropping and vegetative filter strips. Kyalo *et al.* (2014) assessed the benefits of terraces, grass strips, contour farming and cover crops. They reported positive impacts on crop productivity in Lake Naivasha basin, Kenya. Soil and water conservation methods can have an impact on the amount of sediments leaving a catchment and also the amount of water availed to the water users (Lemann *et al.,* 2016). Terraces have been shown to restore soil organic matter by more than a third of that found in areas under conservation agriculture or semi-natural ecosystems (Saiz *et al.*, 2016).

The decision to reduce soil erosion and reverse land degradation should be based on the effectiveness of the soil and water conservation method adopted. Physical determination of the effectiveness of such methods at the farm level is difficult and time consuming. Physics based models can help land planners and policy makers in evaluating the best management practices for use in a given location and also in anticipating changes in ecosystem services with changes in human activities. Hydrological models e.g. SWAT have been used to evaluate conservation methods on sediment and water yield and was also used in this study.

1.2 Problem Statement

The demand for ecosystem services is proportional to increase in the population growth. The increasing population density in Thika-Chania catchment has resulted to intensive cultivation of lands, encroachment of forests, wetlands and riparian areas that are converted to small holder agricultural farms (Vogl *et al.*, 2017). The decline in vegetation cover coupled with steep slopes and high intensity rainfall, has led to increased surface runoff from the degraded areas and agricultural lands resulting to increased soil erosion and subsequent sedimentation in reservoirs. Water quality deterioration have also been observed to increase in the catchment especially during rain seasons. The degradation of water quality has been attributed to the sediments and chemicals carried in surface runoff from agricultural areas and urban centers (Bunyasi *et al.*, 2013). This ultimately influences the costs of water treatment and desilting of water intakes leading to increased cost of water for consumption and hydropower generation.

Lack of adequate water in the rivers especially during dry season has led to insufficient water in reservoirs for supply in Nairobi and other neighboring cities. Inadequate clean water in these areas has been attributed to the outbreak of waterborne diseases. Low flows and sedimentation in reservoirs have also severely affected the hydropower generation leading to power rationing and unstable power supply in the region (Vogl *et al.*, 2017). According to Bunyasi (2012), the reduction of inflow to Masinga reservoir has led to average power output operating below capacity by 16 GWH annually. This in turn affects the hydropower electricity generation and functioning of the Seven Forks power project (Bunyasi, 2012). Without adequate power and declining water quality and quantity, the processes that rely on these services are greatly affected thus influencing the overall economy at large.

Little has been reported on evaluating the existing conservation measures in the catchment and their impacts on water and sediment yield. Inadequate knowledge on the impacts of the conservation measures and lack of incentive programmes have also led to their low adoption rate. This study sought to establish the effects of implementing different soil and water conservation methods on water and sediment yield in Thika-Chania catchment

1.3 Justification

Thika-Chania catchment area plays an important role in Kenya's economy through the ecosystem services it provides. The catchment provides 90% of water consumed in Nairobi and the adjoining towns (Hunink and Droogers, 2015). The major reservoirs supplying this water are Sasumua and Ndakaini which have their main inflow from Chania and Thika river, respectively. Downstream of the catchment are Masinga and Kamburu reservoirs that provide hydropower to majority of the counties. Sagana, Tana, Ndula and Wanji power station are other smaller hydropower stations located downstream of the catchment (Baker *et al.,* 2015). The reservoirs and hydropower stations are greatly affected by sediments affecting their overall production capacity (Bunyasi, 2012; Hunink and Droogers, 2015).

According to Hunink and Droogers (2015), Nairobi City Water and Sewerage Company (NCWSC) indicated that water treatment cost in most cases increase by 33% or more due to sedimentation and disruption of water treatment equipment. The company further spends approximately US$ 50,000 annually for removal of sediments in the intakes works and approximately US$ 188,000 on alum and coagulant in water treatment (Namirembe *et al.,* 2013).

Leisher (2013) recommended the implementation of soil and water conservation measures to control degradation of ecosystem services and hence offset the associated costs. However, prior to their implementation, the impacts of different conservation measures at the catchment and sub-catchment level need to be known. Given that their assessment at the field level could be expensive and time consuming, this study uses a modelling approach to determine the effectiveness of different soil and water conservation measures in the catchment. The results of the study could be used by water practitioners and policy makers to determine what conservation methods are most suitable for implementation at a given catchment or sub-catchment depending on their impacts.

1.4 Objectives

1.4.1 Main objective

To evaluate the impacts of implementing soil and water conservation measures on ecosystem services in Thika-Chania catchment.

1.4.2 Specific objectives

The specific objectives of this study were to;

a) calibrate and validate SWAT model for use in ecosystem services modelling in Thika-Chania catchment.
b) determine the effectiveness of agronomic, vegetative and structural conservation measures on water and sediment yield.
c) evaluate the impact of combining different soil and water conservation methods on water and sediment yield.

1.5 Research Questions

a) What are the optimum parameters values required to calibrate and validate SWAT model for use in simulating water and sediment yield in Thika-Chania catchment?
b) How effective are agronomic, vegetative and structural conservation measures on water and sediment yield?
c) What is the impact of integrating different soil and water conservation methods on water and sediment yield?

1.6 Scope and Limitations of the Study

While this study sought to evaluate sediment and water yield under various soil and water conservation methods, not all possible conservation methods were assessed due to constrain of time and other resources. Only grass strips, terraces and contour farming were assessed within Thika-Chania catchment. Each of the methods was evaluated separately and a combination of the practices were also assessed. The analysis assumed that bench and fanya juu terraces are technically the same and would provide the same ecosystem services if implemented in the catchment. In addition, this study had no direct measurement of sediment and water yield in the field but rather a biophysical model (SWAT) was used. Therefore, the research focused on the biophysical processes and their quantification in terms of ecosystem services (regulating and provisioning services) provided. Further, the study did not look into evaluating the cost effectiveness or the economic valuation of the services.

CHAPTER 2
LITERATURE REVIEW

2.1 Models Application in Assessment of Ecosystem Services

Globally, different models with varying data requirements have been used to evaluate ecosystem services including retention of water and sediments regulation. To determine water and sediment yield from a catchment, seasonal timing of runoff pattern is necessary (Vigerstol and Aukema, 2011). The spatial and temporal runoff variation can be assessed by using a biophysical model. In this section, models that are used simulate ecosystem services are reviewed.

2.1.1 Variable Infiltration Capacity model (VIC)

Variable Infiltration Capacity is a large-scale gridded hydrological model that is most appropriate for large river basins where water yield and stream flow are the main variables of the ecosystem services to be modelled (Liang *et al.*, 1996; Vigerstol and Aukema, 2011). Depending on the data available and the task at hand, the time steps of the model can vary from hourly to daily. The model is suitable for large catchment as the grid cells within which interactions on the land surface are modelled separately are one kilometer or more. The advantage of VIC over the ecosystem specific models like InVEST is that movement of water through different soil layers is accounted for. The model can also account for the variation in water movement as result of soil freezing or melting of the ground ice. The drawback is that a separate routing model is required to combine results from individual pixel and route it to the main channel and eventually through the catchment outlet. The user can determine the output variable to be included in the final results for any time step chosen.

VIC model has been applied in the assessment of large scale hydrological modelling. It has been used to simulate stream flow for basins at a continental level. According to Vigerstol and Aukema (2011), the model has also been used in evaluating land use change on streamflow, assessment of stream flows at global level and also assessing water supply systems in the phase of climate change. The model user comes up with a script to create a range of scenarios to be simulated. VIC however is less useful in comparing various ecosystem services e.g. sediment and pollutant yield from a catchment, that could be assessed while using models like SWAT or ecosystem specific models like InVEST (Francesconi *et al.*, 2016).

2.1.2 Integrated Valuation of Ecosystem Services and Tradeoffs (InVEST)

InVEST model was developed by The Natural Capital Project (www.naturalcapitalproject.org) to address the principles of ecosystem based management. The model combines the working of a biophysical model and the beneficiaries of the services. Tallis and Polasky (2009) argues that in modelling ecosystem services, the supply of the ecosystem services need to be adjusted based on the type, location and intensity of use of each ecosystem service. The value of the ecosystem services that can be modelled using InVEST need therefore to be established.

Depending on the level of use, InVEST has a tiered system of models from tier 1 to tier 3. The level of complexity increases to tier 3. Tier 1 models are freely available online in a software package. Since the modelling of the services is analyzed separately, model results are combined to realize the impacts of changes on the ecosystem services. Tier 1 versions runs on an annual basis contrary to the daily data requirement in SWAT model. This give SWAT a higher confidence on the outputs of the model because it reviews in details the biophysical processes. Therefore in determination of the quantitative estimates of ecosystem services, the predictions of Tier 1 model may be too crude and prone to errors of averaging and aggregation (Tallis and Polasky, 2009). The simplification of the hydrological process by the InVEST model creates a high uncertainty of the results (Vigerstol and Aukema, 2011). The Tier 2 models on the other hard will require more data, have more parameters, time consuming and are difficult to apply (Tallis and Polasky, 2009).

InVEST provides an avenue where one can mix and match Tier 1 and Tier 2 models depending on the data availability and precision required. This way, the model can be customized to suit specific problems. The outputs of InVEST are GIS maps, biophysical service levels and when the valuation option is used, the economic estimates are also given. For instance, in Avoided Reservoir Sedimentation Model, the outputs are the potential and actual average erosion, sediment yield at the outlet, ability of the pixel to reduce erosion and estimated avoided dredge cost.

InVEST can be applied in assessing the change in ecosystem service under different management practices. Most of the InVEST application has been conducted on large scale across the world from prioritization of conservation efforts to simulating land use changes on ecosystems services (Hamel *et al.*, 2015; Tallis and Polasky, 2009; Vigerstol and Aukema, 2011).

2.1.3 Artificial Intelligence for Ecosystem Services (ARIES)

ARIES is an open source, web-based tool used to quantify ecosystem services and their uncertainties by using artificial intelligence techniques to pair locally appropriate data and the model (Bagstad *et al.,* 2013; Villa *et al.,* 2009). The users gather most of the local data available and then the model uses probabilistic Bayesian networks to establish relationships between input data and the ecosystem service values. In areas where adequate local data is available, ARIES can use the biophysical relationships to model the ecosystem services. The user can take the data stored within the tool to do a random assessment of ES with wider range of uncertainties. Detailed data plus local knowledge is required to build the linkage between the input data and outputs of the ecosystem service and improve the range of uncertainty. ARIES has the capability to cover flood control, sediment regulation, water supply among other services. Different scenarios can be assessed and valuation of the services determined as in InVEST model. The outputs of the model are indicated as a range that shows the confidence level of the results. However, sediment yield is not explicitly provided but rather combined into reporting on results e.g. economic valuation (Vigerstol and Aukema, 2011).

ARIES model have been used to analyze catchment biophysical processes and the valuation of ecosystem services (Bagstad *et al.,* 2013; Hamel *et al.,* 2015; Villa *et al.,* 2009). Hamel *et al.* (2015) modeled the sediment retention service in a Cape Fear catchment in North Carolina, USA and found that spatial variability of the sediment prediction was well represented in the model. They also reported that the model fairly simulated ecosystem services without calibration. Bagstad *et al.* (2013) compared ARIES and InVEST the model in a case study in San Pedro River, Arizona. They concluded that the results were closely aligned for landscape-scale urban-growth scenarios and different for mesquite-management scenarios. According to Vigerstol and Aukema (2011), the lack of transparency due to complexity of the model code hinders the communication of model relationships and results to the stakeholders.

2.1.4 OPAL (Offset Portfolio Analyzer and Locator).

OPAL evaluates and quantifies the impacts of development on ecosystem services and facilitate the selection of measures to offset the losses. Model users define the spatial data and information related to the beneficiaries. OPAL is able to quantify the impacts of alternative land uses on ecosystem services based on the kind of input data provided. The tool then filters offset sites based on the modelled impacts and the provided priorities and preferences by the user thus getting rid of the sites that do not meet the set requirements (Mandle *et al.*, 2016). The tool ultimately gives an output of the results in an interactive format for interpretation.

2.1.5 Multiscale Integrated Models of Ecosystem Services (MIMES)

MIMES are public domain, spatially explicit tools that biophysically model the ecosystem services. However, the model has not been widely documented compared to SWAT or InVEST tools. MIMES is a dynamic model that was developed to account for temporal dynamics and incorporate existing ecological process models into ecosystem modelling and finally evaluate the value of the ecosystem services economically. The model uses a commercial coding and simulation software package called Simile (Bagstad *et al.,* 2013). Input data required in setting up the model include spatial data and information required to perform model parameterization. Significant amount of time is also consumed in setting up the model. The output of the model consist of time series of ecosystem service values (Bagstad *et al.,* 2013).

2.1.6 Soil and Water Assessment Tool (SWAT)

SWAT is a physically based, continuous time model that requires weather data, topography, soil properties and land use information to assess the impact of land management practices on water, sediments and agricultural chemicals yield (Neitsch *et al.,* 2005). Using this input data, the physical processes of water and sediment movements are modelled directly by SWAT on a daily time step at the sub-catchment level. Different scenarios are assessed by simulating land use change in the SWAT model (Vigerstol and Aukema, 2011). SWAT results include aggregated data on monthly and yearly basis.

The model has been used across the world to evaluate different ecosystem services (Cho *et al.,* 2010; Hunink *et al.*, 2012; Jha *et al.,* 2004; Parajuli *et al.,* 2008; Tuppad *et al.,* 2010). For instance, Cho *et al.* (2010) evaluated the effects of filter width in coastal plains of Georgia using SWAT model while in Thailand it has been used to model the impacts of different land management options in Lam-Sonthi catchment (Phomcha *et al.,* 2012).

2.1.7 Model selection

Tools that are well tested and documented can add credibility and trust to the decision process and increase the confidence of the stakeholders in their use (Bagstad *et al.,* 2013). The scope of this research was to evaluate the effectiveness of soil conservation measures on water and sediment yield. A model that is physics-based, spatially explicit, deterministic (considering the biophysical processes) was required. Sediments yield is among the outputs expected in this study hence the VIC model was not suitable. SWAT modelling is conducted on a daily time step compared to the annual time step in InVEST and thus SWAT guarantees more reliable scenario simulations. The model focuses on water and erosion processes as influenced by different methods of land management. Besides, SWAT has been used extensively in modelling ecosystem services in African countries and across the world (Hunink *et al.*, 2013; Jha *et al.*, 2004; Kigira *et al.*, 2010; Lemann *et al.*, 2016; Motsinger *et al.*, 2016; Mwangi *et al.*, 2015; Phomcha *et al.*, 2012; Silvestri *et al.*, 2013). According to Francesconi *et al.* (2016), provisioning and regulatory services are easily quantifiable in SWAT.

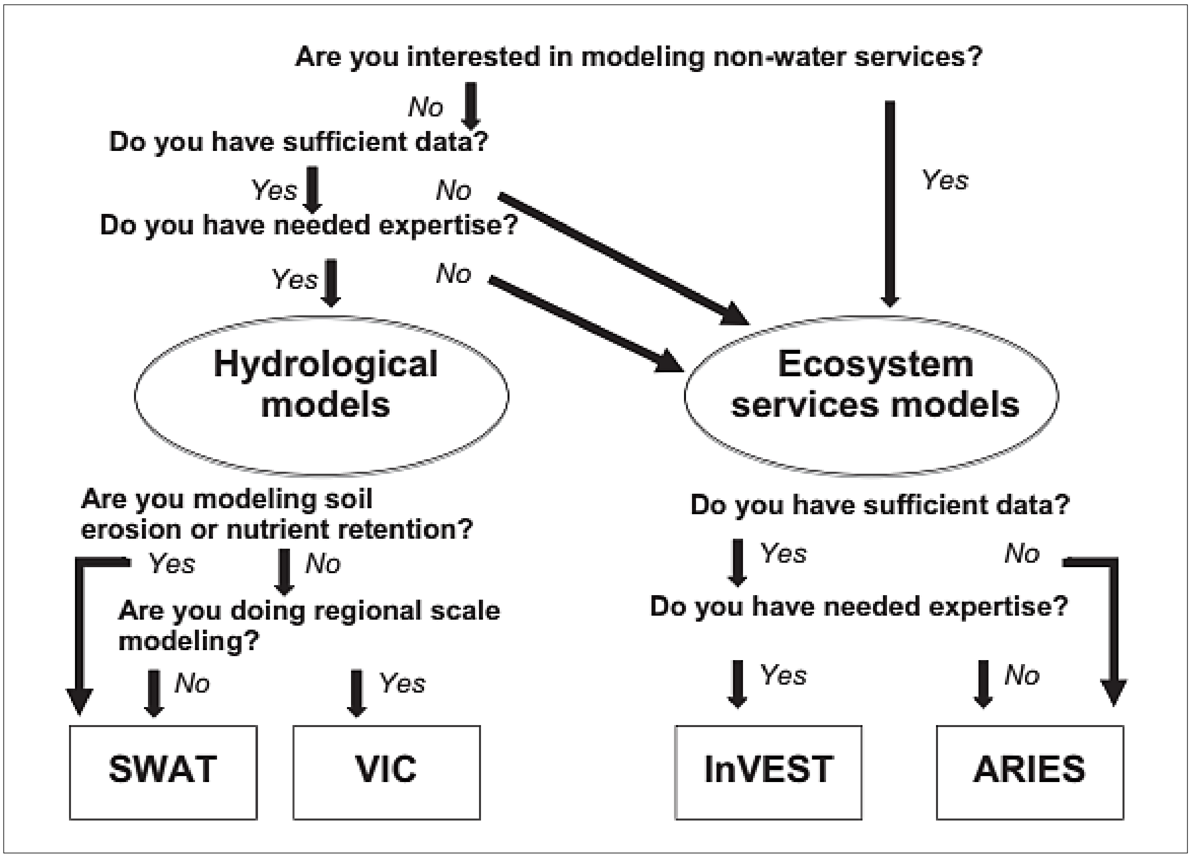

Figure 2.1 Schematic for model selection (Vigerstol and Aukema, 2011).

Other factors considered in model selection for use in this study include data requirement, model quantification and uncertainty, capacity for independent application, level of development and documentation, generalizability and level of knowledge/expertise required (Figure 2.1). The perception and views of the stakeholders on the accuracy of a model in simulating different scenarios influences its credibility. Vigerstol and Aukema (2011) recommended the use of SWAT model where peer review and scientific consensus of the underlying biophysical processes is needed. SWAT model was therefore selected to evaluate the impacts of soil and water conservation method on ecosystem services.

2.1.8 Overview of the SWAT model

SWAT is a process based distributed model that operates on a daily time step to model ecosystem processes. The tool is mainly composed of hydrological component, weather, soil properties, plant growth, nutrients, bacteria and land management.

In SWAT modelling, the catchment is subdivided into several sub-catchments that are determined by a user defined Critical Source Area (CSA) which is the minimum area the model requires for the channel process initiation. The CSA determines the number of sub-catchments produced in a catchment delineation process and also channel networks density. Research has shown that in the prediction of sediment yield, a sub-catchment should be approximately 2% - 5% of the total catchment area. The sub-catchments are the then divided into Hydrologic Response Units (HRUs) each of which represents discrete land use, soil attributes and slopes. The HRUs are represented as a percentage of the sub basin area in SWAT and may not be identified spatially (Arnold *et al.,* 2012).

The biophysical processes in SWAT are driven by the water balance due to its influence on plant growth and transport of sediments and nutrients catchment. Hydrology in a catchment is divided into two phases; the land phase and the routing phase (Arnold *et al.,* 2012). The land phase controls the amounts of sediments, water, pesticides and nutrients that enter the main channel for every sub-catchment. Water and sediments movement through the river or channel is controlled in the routing phase (Neitsch *et al.,* 2005). In the SWAT model, the land phase is described by the following equation;

$$SW_t = SW_0 + \sum_{i=1}^{t}\left(R - Qs - E_a - W_{seep} - Q_{gw}\right) \tag{2.1}$$

Where SW_t is the final soil water content (mm), SW_o is the initial water content in day *i* (mm), R is the amount of precipitation in day *i* (mm), Q_s is the amount of surface runoff in day *i* (mm), E_a is the amount of evaporation in day *i* (mm), W_{seep} is the amount of water entering the vandose zone in day *i* (mm) and Q_{qw} is the amount of return flow in day *i* (mm).

After determination of sediments, nutrients and water loading from the land phase to the main channel, the loadings are routed through the rivers and reservoirs in the catchment. Among the hydrological processes modelled in SWAT include surface runoff, infiltration, evapotranspiration, recharge etc. The tool also uses a pant growth model to simulate land cover and differentiate between annual and perennial plants (Arnold *et al.,* 2012).

2.1.9 Sediment routing

Deposition and degradation are main process that control the movement of sediment in the channel. Williams (1980) simplified version of Bagnold's (1977) stream power definition is used in SWAT to develop a way of determining channel degradation as a function of channel slope and velocity. The maximum amount of sediments that can be transported from a channel segment is simulated as a function of peak flow rate as shown in equation (2.2).

$$S_{max} = C_{sp}\left(\frac{q_s}{A_{ch}}\right)^{sp} \qquad (2.2)$$

Where S_{max} is the maximum sediment concentration transported or the channel carrying capacity (10^3 kg/m^3), c_{sp} is an empirical coefficient that needs to be calibrated, q_s is the peak flow rate (m^3/s), A_{ch} is the cross-sectional area of flow in the channel and sp is a user defined exponent.

Sediment routing in streams is estimated by comparing available sediment load and estimated transport capacity in a given stream segment (Cho *et al.,* 2010). Sediment yield in each HRU is routed to the channel of the corresponding sub-catchment. In scenarios where the transport capacity is larger than sediment load, degradation of the channel will occur. On the other hand, if the sediment load in a channel segment is larger than the transport capacity, then deposition of sediments will occur within the stream as exemplified in equation (2.3).

$$sed_{deg} = (S_{max} - S) \times V_{ch} \times K_{ch} \times C_{ch} \qquad (2.3)$$

Where sed_{deg} is amount of sediment re-entrained in channel segment (tons), $V_{ch}$is the volume of water in the channel segment (m^3), K_{ch} is the channel erodibility factor (cm/hr/Pa) and $C_{ch}$is the channel cover factor.

Surface runoff in SWAT is estimated using either the SCS curve number or Green-Ampt infiltration (Neitsch *et al.,* 2005). In this research, the SCS curve number was used as shown in equation (2.4). The SCS curve number is a function of land use, soils permeability and the antecedent soil water conditions.

$$Q_{surf} = \frac{\left(R_{day} - I_a\right)^2}{\left(R_{day} - I_a + S\right)} \tag{2.4}$$

Where, Q_{surf} is the accumulated runoff or rainfall excess (mm), R_{day} is the rainfall depth for the day (mm), S is the retention parameter (mm), and I_a is the initial abstractions which includes surface storage, interception and infiltration prior to runoff (mm). The retention parameter, S is defined in equation (2.5).

$$S = 25.4\left(\frac{100}{CN} - 10\right) \tag{2.5}$$

Where CN is the curve number of the day and Ia is approximated as 0.2 S and therefore the surface runoff (Q_{surf}) can be computed as;

$$Q_{surf} = \frac{\left(R_{day} - 0.2S\right)^2}{\left(R_{day} + 0.8S\right)} \tag{2.6}$$

Runoff occurs only when the rainfall depth for the day (R_{day}) is less than the initial abstractions.

2.1.10 Sensitivity analysis, calibration and validation of SWAT model

Parameters used in the SWAT model are process based and must be within an acceptable uncertainty range. Sensitivity analysis, which is the process of determining the rate of change in the outputs of the model with respect to changes in the model parameters, must be conducted before calibration and validation (Winchell *et al.*, 2013). In modelling of sediment and water yield, the predominant and most sensitive parameters need to be identified, calibrated and validated. Sensitivity analysis can be local or global; in local analysis, only one parameter is allowed to change while in global analysis all parameters are allowed to change (Abbaspour, 2015). Global sensitivity analysis requires a wide range of simulations while in local sensitivity analysis, it is uncertain how other parameters change. Parameter sensitivities are determined by calculating a multiple regression system which regresses the Latin hypercube generated parameters against the objective function values as illustrated in equation (2.7).

$$g = \alpha + \sum_{i=1}^{m} \beta_i b_i \tag{2.7}$$

A t- test, which is the coefficient of the parameter divided by its standard error, is then used to identify the relative significance of each parameter, b_i. It is used to determine the precision with which the regression coefficient is measured.

Calibration is the process of reducing the model prediction uncertainty by adjusting the parameters of the model to a certain local condition (Abbaspour, 2015; Neitsch *et al.,* 2005). Model calibration is then achieved by comparing model predictions for a given set of assumed conditions with observed data for the same conditions (Arnold *et al.,* 2012). After calibration, the variable of interest is then validated; in this case streamflow and sediment yield. Validation is the process of making sure that the model can produce results that are satisfactorily accurate (Neitsch *et al.,* 2005). The parameters that were adjusted during calibration process are used to predict the observed data selected for the validation process.

Statistical methods are used to compare the best fit between the observed and the simulated values; they include but not limited to Nash-Sutcliffe Efficiency (NSE), Percent BIAS (PBIAS) and coefficient of determination (r^2). The r^2 ranges from 0-1 where 0 indicates that are no correlation and 1 indicates perfect correlation between the observed values and model predictions. The NSE values on the other hand ranges from $-\infty$ to 1. The NSE provides an indication of how perfect the model output matches the observed data. A perfect match between the observed and simulated values is indicated by a value of 1. Any value less than 0 signifies that the observed data mean is more accurate predictor than the model output.

SWAT calibration can be manual or semi-automatic. A semi-automated calibration and validation is achieved using a calibration and validation programme called SWAT Calibration and Uncertainty Program (SWAT-CUP) (Abbaspour, 2015). In calibration and validation, the available observed data are split into two datasets where one set is used for calibration and another for validation (Arnold *et al.,* 2012). The climate data used for both calibration and validation should be substantially different. The wet, moderate and dry periods, however, should occur in both calibration and validation datasets. This is achieved by ensuring that the data are split frequently by time periods considering all the events in both periods (Neitsch *et al.,* 2005). In some situations, data may not be enough to split them into two for calibration and validation. In such cases, data can be split spatially. Data from a given station is used in the calibration phase while dataset in a different monitoring station are used for validation (Neitsch *et al.,* 2005).

2.1.11 SWAT Calibration and Uncertainty Programs (SWAT-CUP)

SWAT-CUP is a software that incoprorates the automatic calibration, validation and uncertainty analysis in SWAT model. All SWAT parameters can be included during the calibration process including management, basin and weather generator parameters. SWAT-CUP includes a multi-site, semi automated inverse modelling routine SUFI2 for calibration and uncertaintiy analaysis (Faramarzi *et al.*, 2009).

In SUFI2, uncertain parameters accounts for all sources on uncertainties including climate data, model parameters and observed data. Propagation of the uncertainties in the parameters leads to uncertainties in the model output which is expressed as 95% probability distributions (Abbaspour, 2015). Propagation of uncertainties is conducted using Latin hypercube sampling expressed as 95% prediction uncertainty (95 PPU). These are the model ouputs in a stochastic calibration approach. Measures quantifying the strength of a calibration or uncertainty analysis include the r-factor which is the average thickness of the 95 PPU band divided by the standard deviation of the measured data. Calibration and prediction uncertainty is judged on the basis of the closeness of the p-factor to 100% (i.e., all observations bracketed by the prediction uncertainty) and the r-factor to 1. The r-factor is determined using equation (2.8).

$$\mathrm{r-factor} = \frac{\mathrm{p-factor}}{\sigma_{\mathrm{obs}}} \tag{2.8}$$

Several iterarions can be used in SUFI2 program where in each iteration, parameters ranges get closer to the region of the parameter space which provided better results in the previous iteration. As the parameter range becomes smaller, the 95PPU envelope get smaller meaning that the objective function gets better in the subsequent iterations.

2.1.12 Universal Soil Loss Equation (USLE)

The Universal Soil Loss Equation (USLE) was designed to calculate average soil loss over a long-time from rill and sheet erosion and under given conditions (Wischmeir and Smith, 1978). The amount of soil generated is usually a product of six factors that are related to rainfall, soil, topography, land cover and land use/management as shown in equation (2.9).

$$A = R \times K \times LS \times C \times P \tag{2.9}$$

Where; A = the mean annual soil loss (tons/ha/yr.), R = rainfall – runoff erosivity factor, K = soil erodibility factor, LS = topographic factor, C = crop cover and management factor, P = support practice factor.

Rainfall runoff erosivity factor is computed as product of the total storm energy (E) and the maximum 30-minute intensity of rain (I_{30}). The erosivity factor corresponds closely with the amount of soil that is lost from the field (Morgan, 2005). Soil erodibility factor refers to the susceptibility of the soil to erosion and is affected by the physical, chemical and morphological components of the soil (Humberto and Rattan, 2008).

The topographic factor (LS) is computed as the ratio of soil loss from a given soil to that from saturated USLE plot of 22.1 m length and having a 9% slope as illustrated in equations (2.10).

$$LS = \left(\frac{Length}{22.1}\right)^{m} (65.41\, sin^2\theta + 4.56\, sin\theta + 0.065)$$

$$m = 0.6[1 - \exp(-35.835 \times S)] \quad (2.10)$$

$$\theta = tan^{-1}\left(\frac{S}{100}\right)$$

Where, S is the slope of the field in percentage and θ is the steepness of the field in degrees. The slope length indicated in SWAT as *SLSUBBSN* is given as;

$$SLSUBBSN = (x \times SLOPE + y) \times \frac{100}{SLOPE} \quad (2.11)$$

Where x is a dimensionless variable with values from 0.12-0.24, y is a dimensionless variable that is influenced by soil erodibility, crop management system, crop system and *SLOPE* is the average slope of the field. The variable y can take values of 0.3, 0.6, 0.9 or 1.2 (Arabi, Frankenberger, Engel, and Arnold, 2008). The low value of 0.3 is used for highly erodible soil with conventional tillage and little residue, while the high value of 1.2 is used for soil with very low erodibility and with a residue of more than 3.3 t/ha or no-till management condition. The impact of the slope length on the erosion can be estimated by equation (2.12) (Arabi *et al.*, 2008);

$$Sed \propto SLSUBBSN^{(m-0.336)} \quad (2.12)$$

Where, *Sed* is the sheet erosion computed for the HRU. The value of m increases with increase in the slope of the HRU. Therefore, the higher the slope, the more the sediment yielded from a given area.

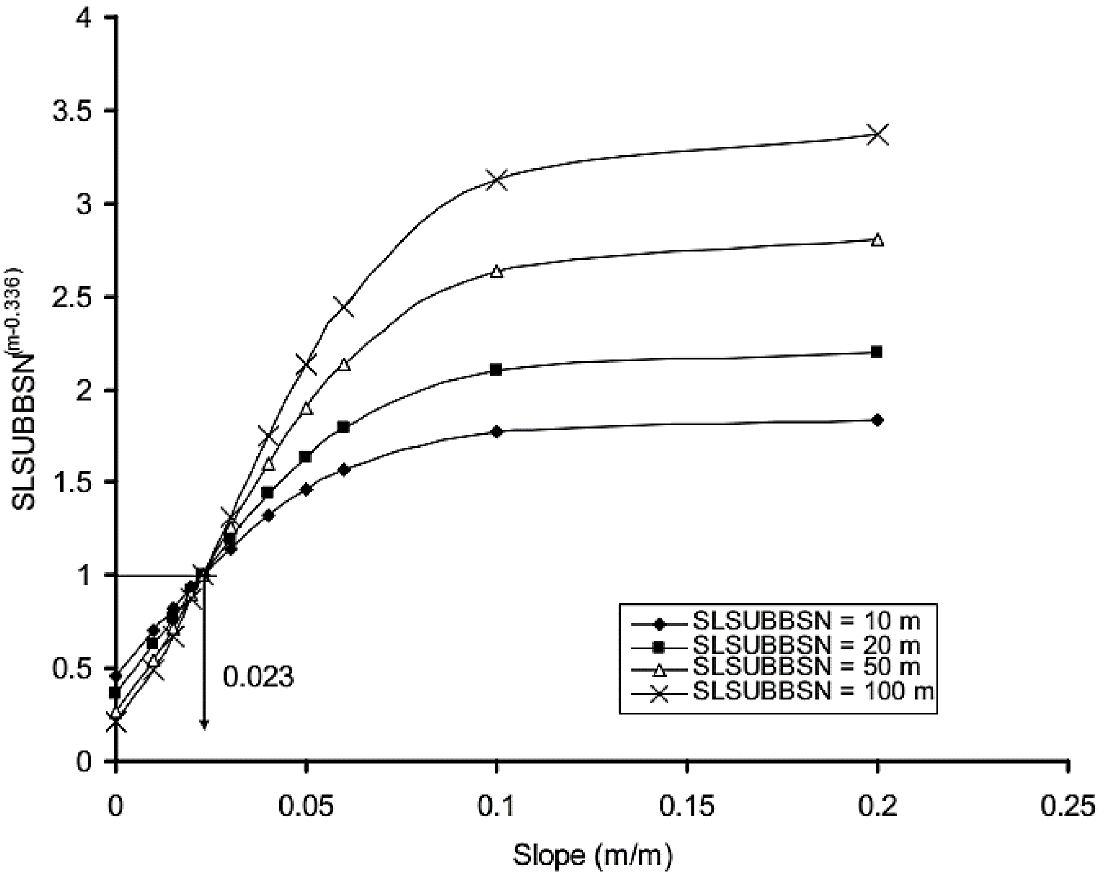

*Figure 2.2 Relationship between slopes and soil loss (*Arabi *et al.*, 2008).

From Figure 2.2, SWAT estimated upland erosion is inversely correlated to the slope length in areas where *m* is less than 0.336 or where slope of the HRU is less than 2.3%. In this case, reducing the slope length will result to higher erosion estimates.

Crop cover management factor (C) is determined as the ratio of soil loss from a field under a given crop stage period compared to the soil loss in a field under continuous bare fallow where tillage is up and down slope (Humberto and Rattan, 2010). It is the ratio of soil loss from a land cropped under specified conditions to the corresponding loss from a clean tilled, continuous fallow (Wischmeir and Smith, 1978). It infers that soil loss changes is influenced by vegetative crop cover during the different crop stages like rough fallow, seedling, establishment, growing, maturing, residue and stubble (Humberto and Rattan, 2008). The estimates of crop cover management factor can be found in literature for the different vegetation cover.

Support practice factor (P) are the conservation methods or practices that are used to reduce soil erosion (Morgan, 2005). It is determined as the ratio of soil lost in a field with support practices to the soil loss from a field under up and down cultivation without practices. The values of P ranges from 0 to 1 where a value of one denotes a bare soil without any support practices. In systems with various support practices, for instance contouring, strip cropping and terracing, the P values can be calculated as shown in equation (2.13).

$$P = P_c \times P_s \times P_t \tag{2.13}$$

Where, P_c is the contouring factor, P_s is the strip cropping factor and P_t is the terracing sedimentation factor.

2.1.13 Modified Universal Soil Loss Equation (MUSLE)

Erosion calculation and sediment yield calculation in SWAT uses the Modified Universal Soil Loss Equation (MUSLE) which is a revised version of the USLE. MUSLE predicts sediment yield based on runoff factor while USLE predicts sediment yield based on rainfall energy factor (Humberto and Rattan, 2008; Neitsch *et al.,* 2005). MUSLE therefore accounts for the antecedent soil moisture and estimates of sediment from a single storm. The model therefore improves sediment yield prediction and eliminates the need for delivery ratios because the runoff factor represents the energy used in detachment and transport of sediments (Neitsch *et al.*, 2005).

$$Sed = 11.8 \left(Q \times q_p \times A\right)^{0.56} \times K \times C \times P \times LS \times CFRG \tag{2.14}$$

Where, *Sed* is the sediment yield from a given HRU on storm event basis (tons/ha), Q is the surface runoff volume (mm/ha), q_p is the peak runoff (m^3/s), K is the USLE erodibility factor, C is the USLE crop and management factor, LS is the USLE topographic factor, A is the Hydrologic Response Unit (HRU) (ha) and CFRG is the coarse fragment factor estimated as;

$$CFRG = \exp(-0.053 \times Rock) \tag{2.15}$$

Where, *Rock* is the % rock in the upper most soil layer.

To model the peak runoff rate in SWAT, the modified rational method, equation (2.16), is used

$$q_{peak} = \frac{\alpha_{tc} \times Q \times A}{3.6 \times t_c} \tag{2.16}$$

Where, q_{peak} is the peak runoff rate (m/s); Q surface runoff (mm), α_{tc} is fraction of the daily rainfall that occurs during the time of concentration (t_c), A is the area of the sub-catchment in km^2. Time of concentration is given as;

$$t_c = \frac{SLSUBBSN^{0.6} \times OV\_N^{0.6}}{18 \times SLOPE^{0.3}} \tag{2.17}$$

Where, *SLSUBBSN* = is the slope length (m), *OV_N* is the Manning's roughness coefficient and *SLOPE* is slope of the sub-catchment (m/m).

2.2 Ecosystem and Ecosystem Services

According to the Millennium Ecosystem Assessment, MEA (2005)**,** an ecosystem is a dynamic complex of the living and non–living environment that interact as a functional unit. An ecosystem represents a collection of plants, animal and microorganisms interacting with each other and the nonliving components of the environment (WRI, 2008). Ecosystem services on the other hand are the benefits that people obtain from the ecosystem (Kauffman *et al.,* 2014; Millennium Ecosystem Asessment, 2005). They include a set of ecosystem functions that are useful to the people (Kremen, 2005).

The ability to have adequate and clean drinking water, access to resources to earn income and improve livelihoods, ability to reduce vulnerability to ecological shocks, are some of the determinants and constituents of human well–being (Millennium Ecosystem Asessment, 2005). The ecosystems services include the regulating, provisioning and cultural services that directly influence people and supporting services that are required to maintain the functioning of the other services (Zhang *et al.*, 2007).

Provisioning services include such products or benefits that are derived from the ecosystems e.g. food, fresh water, building materials, fuelwood, bio–chemicals etc. Regulating services on the other hand are the good values that people get from the regulation of ecosystem processes as presented in Figure 2.3.

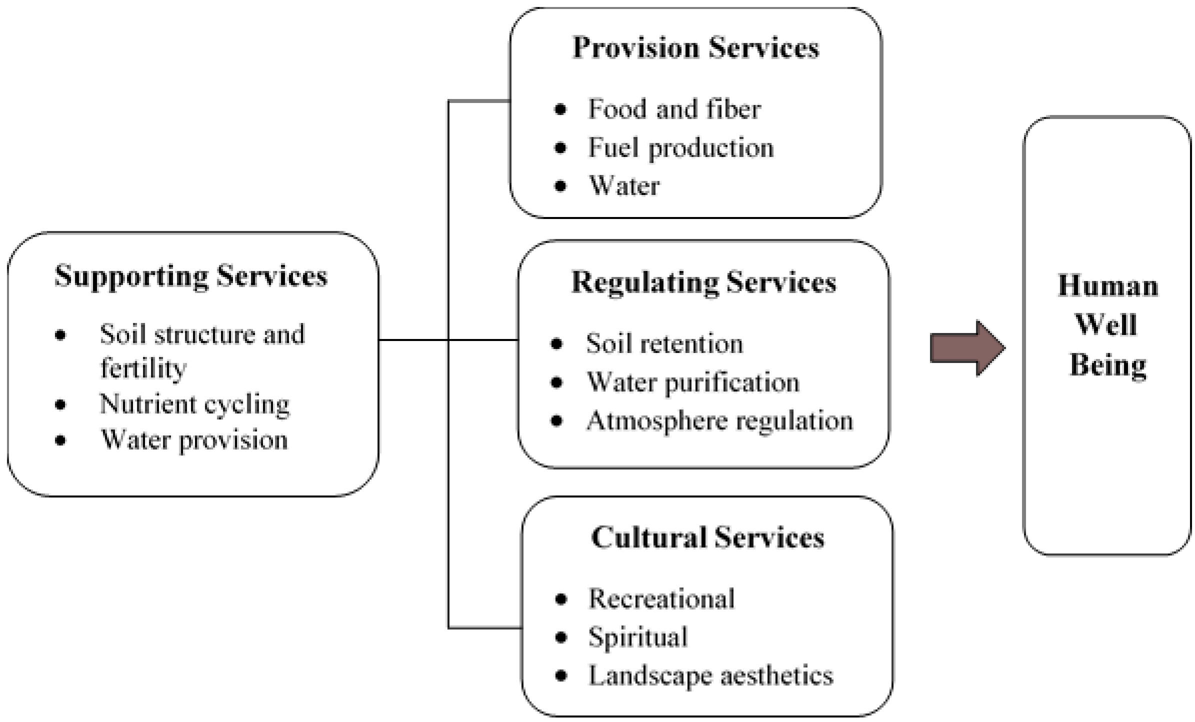

Figure 2.3 Classification of ecosystem services

Cultural services include the non–material benefits e.g. recreation and landscape aesthetics that people obtain from the ecosystems. Supporting services are necessary for the production of all other ecosystem services e.g. nutrient cycling, photosynthesis/primary productivity and soil formation. The analysis of supporting and cultural ecosystem services is less common probably because they are not directly utilized by people yet they are necessary for proper functioning of the ecosystem (Francesconi *et al.,* 2016).

Across the globe, studies have focused mainly on more than one ecosystem services to evaluate the biophysical processes and economic valuation of the catchment (Wangai *et al.,* 2016). Ajwang' *et al.* (2016), assessed the provisioning and cultural ecosystem services in natural wetlands and rice fields in Kenya. They evaluated various ecosystem services including fiber, building materials and food stuff that were quantified either as biophysical quantities or monetary value. They demonstrated that the provisioning services of the ecosystem have significantly declined in the past decades.

To achieve better productivity and sustainable livelihoods for human needs, adequate and sustainable supply of multiple ecosystem services is required. However, these services have been threatened by anthropogenic activities resulting from increased population leading to unsustainable land use and production systems (Wangai *et al.,* 2016). The outcome is an intensive cultivation of lands and increased felling of trees that accelerates the deterioration of the ecosystem services (Millennium Ecosystem Asessment, 2005).

Extreme events resulting from changing climate has been shown to influence climate change (Zommers *et al.,* 2016). Floods, for instance, would cause soil erosion and landslides that affects the water quality and crop productivity when fertilizers and pesticides are washed away in surface runoff. Temperature increase and decreased rainfall leads to reduced crop yield and vegetation cover which ultimately decreases the capacity of ecosystems to trap sediments and regulate surface runoff (Carpenter *et al.*, 2006; Zommers *et al.*, 2016). Carpenter *et al.* (2006) observed that the reliability and flow of ecosystem services can be altered by ecological changes that in the end increase the vulnerability of people to further changes. Efforts to improve the ecosystems services is highly encouraged to improve food production, water quality, reduce soil erosion and sedimentation. Studies in Kenya have indicated a decline in the ecosystem services and have recommended intervention measures to restore their functionality (Wangai *et al.,* 2016). Conducting studies on the effectiveness of such measures that enhance the provisioning and regulatory services will help in the management of the catchment. According to the Millennium Ecosystem Asessment (2005), stakeholders have become more involved in the decision-making process but the challenge is the provision of adequate and precise information.

2.2.1 Payment for ecosystem services

The World Overview of Conservation Approaches and Technologies (WOCAT) has listed methodologies to prevent soil erosion and improve water management. However, these methods are not widely applied owing to the fact that farmers lack the ability or willingness to implement them (WOCAT, 2007). According to Kauffman *et al.* (2014), if the suggested methods are not supported financially, their adoptability by farmers and other stakeholders is compromised. Payment for Ecosystem Services (PES) is an economic tool where there are sellers and buyers of the services provided (Smith *et al.,* 2013). The buyers provide the incentives to the sellers (usually land users) mainly to encourage them continue supplying ecosystem services.

PES has become increasingly significant in developing contries due to its potential to contribute to the sustainable use of natural resources and poverty reduction (Sand *et al.,* 2014). Monetary valuation and payment schemes has attracted support from cross cutting stakeholders including governments for conservation and commodifying the ecosystem services (Gómez-baggethun *et al.*, 2010). PES can provide environmental services such as water quality improvement and regulated flow of water during the dry seasons when implemented at the catchment or sub-catchment level.

A study was conducted by Nduhiu *et al.* (2016) in Sasumua catchment to determine the influnce of payment for ecosystem services on ecosystem services. They reported that the total supended solids (TSS) was reduced from a baseline average of 71.05 mg/L to 42.73 mg/L after one year of PES implementaion. Therefore, introducing PES as an incentive program for practicing soil and water conservation in catchment would improve the quality of water downstream and also reduce the amount of fertilizers, sediments and nutrients carried in runoff. The results of the present study can providedecison makers with information on the methods that are effective in improving water quality and reducing sediment yield from the catchment for application in PES programs.

2.2.2 Soil erosion

Globally, water erosion is the most severe type of soil erosion that is common in humid and sub-humid regions while wind erosion is mostly common in arid and semi–arid regions (Angima *et al.*, 2003; Humberto and Rattan, 2008). Erosion by water occurs in the form of sheet, rill, splash and gully erosion (Morgan, 2005). Globally, erosion associated with agricultural land is more widespread compared to any other land use type (Humberto and Rattan, 2008; Morgan, 2005). Soil erosion has its long-term effects on soil productivity, sustainable agriculture and environmental damage through sedimentation, water quality degradation and increased flooding. The implication is that soil erosion is a contributing factor to productivity loss of ecosystem services through gradual degradation (Humberto and Rattan, 2008; Kyalo *et al.*, 2014).

Soil erosion prevention entails reducing the rate of soil erosion to match approximately to the erosion under natural condition (Morgan, 2005). This can be achieved by using soil and water conservation methods. Mati *et al.* (2000) assessed erosion hazards in Kenya and found that improper land use and management were the major factors associated with soil erosion. In order to select the best conservation methods for soil erosion prevention and control, an understanding of the processes and influencing factors ought to be understood.

2.3 Agronomic and Vegetative Conservation Measures

Soil and water conservation involve using the land within the limits of economic practicability protecting it from degradation or depletion by erosion, plant nutrient exhaustion, waterlogging, or unsustainable cultivation (CARDI, 2010).

Agronomic and vegetative conservation measures are soil management practices that aim at decreasing the rainfall erosivity, increase water infiltration, decrease the amount of runoff and lessen soil movement (Humberto and Rattan, 2008; Morgan, 2005). They include the soil conservation measures undertaken at the farm level to improve the productive capacity of the land (WOCAT, 2007). Agronomic conservation measures are the activities undertaken within the cropping area for production of crops and they include such practices as intercropping, contour farming, minimum tillage and mulching. Agronomic measures are normally associated with annual crops and are practiced routinely every planting season or in rotation (Mwangi *et al.,* 2013). Vegetative conservation measures include such practices as filter strips, border strips and strip cropping that trap sediments, fertilizers, and bacteria in surface runoff (Arabi *et al.,* 2008)

2.3.1 Contour farming

Contour farming is an agronomic conservation method where farming activities are conducted across the slope (Morgan, 2005). The practice reduces soil erosion by creating crop row ridges that act as barriers to surface runoff and therefore reducing the velocity and enhancing infiltration of water in the small depressions formed (Arabi *et al.,* 2008; Stevens *et al.,* 2009). Farming along the contours provides almost complete protection against erosion from storms of low to moderate intensity and is most effective on slopes of 3 to 8% (Neitsch *et al.,* 2005; NRCS, 2007). Ultimately, the method help in reducing sedimentation in reservoirs and also improving the soil and water quality (Branca *et al.,* 2011; NRCS, 2007).

Contour cultivations have been adopted across the world as an effective way of reducing soil erosion and prevent sedimentation (Quinton and Catt, 2004; Tadesse and Morgan, 1996; Yuan *et al.,* 2014). Quinton and Catt (2004) studied the effects of cultivation up and down slope and found that the mean runoff from a 1.32 mm rainfall event was higher by 0.82 mm than that from contour farming. In Uganda, a study was carried out to evaluate the influence of different land management methods on soil erosion where soil loss for individual soil landscape units were modelled on a hillslope using Water Erosion Prediction Project (WEPP) model. In that study, Brunner *et al.* (2008) found that soil loss reduced by 13% from the base scenario value when contour farming was practiced.

Simulations using SWAT model have been used to assess the impact of Best Management Practices (BMPs) including the use of contours on sediment yield and nutrient loss. Tuppad *et al. (*2010) evaluated the impact of contour farming in Bosque River catchment in Central Texas and concluded that the practice reduced sediment loss by a mean of 28% to 67% from the base scenario at the HRU level. The authors also observed that the practice factor used greatly influences the effectiveness of the soil conservation method used. For example, when they used a P–factor of 0.5, the reduction of total phosphorous was 11.6% but with a P–factor value of 0.4, the reduction increased to 14.1%. Therefore, a careful choice should be made when determining the value of the parameters used to represent a particular conservation method

Contour hedgerows consisting of various combinations of tree and grass can be used on sloping lands to minimize erosion and improve crop productivity by restoring the fertility of the soil (Angima *et al.,* 2002; Geertsma *et al.,* 2011). Angima *et al.* (2002) studied a cumulative data of five cropping seasons and reported that contour hedges on 20% slope conserved 168 Mg/ha and 146 Mg/ha on a 40% slope using a P-factor of 0.7. They concluded that contour farming can be used by local farmers that use mixed farming in Central Kenya in which Thika-Chania catchment lies.

Farmers decision to implement soil and water conservation measures is greatly influenced by size, topography and soil types of their land (Kyalo *et al.*, 2014). Contour farming has been shown as among the soil and water conservation practices in the catchment (Leisher, 2013) Therefore, the impact of contour farming on ecosystem services in the catchment need to be quantified to inform stakeholders on its effectiveness.

Contour farming has been shown to have drawbacks based on where they are implemented and the level of farm mechanization in place. Quinton and Catt (2004), observed that on steep slopes, water can accumulate in lowest point of the contours and eventually break through to form large rills. Movement and operation of agricultural machinery across the slope is also challenging on steep slopes.

2.3.2 Filter strips

According to USDA-NRCS (2016), a filter strip is an area of herbaceous vegetation that removes contaminants from overland flow. The main purpose of filter strips is to reduce the amount of suspended materials and sediments in runoff water (Tuppad *et al.,* 2010). This ultimately helps in maintaining and improving the ecosystem services. Filter strips can be implemented in critical areas like along the rives edges and reservoirs that need to be protected from sediments and other contaminants including fertilizers and pesticides from agricultural fields. Experimental plots and use of models to evaluate effectiveness if filters strips on ecosystems have been studied across the world (Cho *et al.,* 2010; Droogers *et al.,* 2011; Herweg and Ludi, 1999; Parajuli *et al.,* 2008; Vogl *et al.,* 2016). However, evaluation of impacts using experimental methods can be time consuming and expensive compared to model simulation.

Contaminants removal from runoff and the effectiveness of vegetative strip in trapping sediments depends on the width of vegetative strips (Yuan *et al.,* 2014). There are other factors that affect the trap efficiency of the vegetative strips like the inflow rate of the runoff, the slope steepness and length, and the size of the transported particles (Akan *et al.,* 2014). According to USDA-NRCS (2016), a well installed filter strip can effectively trap up to 90% of sediment and nutrients from the land.

Sediments and pollutants reduction rate in SWAT model is estimated as a function of average width of the filter strip and water volume from the stream segment (Cho *et al.,* 2010). The filter width is represented in the model as a parameter indicated as FILTERW. However, this simulation is limited to the fact that it does not consider the spatial distribution of the filter strips across the catchment (Cho *et al.,* 2010). Filter strips modelling is influenced by value of the Critical Source Area used. Cho *et al.* (2010) assessed the effects of catchment sub-division and filter width on SWAT simulation and found that the total sediments yield at the catchment outlet were influenced by the characteristics of channel segment in the outlet sub-catchment which are influenced by CSA.

Tuppad *et al.* (2010), simulated agricultural management alternatives for catchment protection in Bosque River catchment in Texas and found that filter strips reduced sediments depending on their widths. For example, at 2 m and 12 m filter strips width, Tuppad *et al.* (2010) estimated reduction in sediment load as 7% and 13%, respectively from the baseline scenario value. The authors recommended further testing of the filter strips in other geographical settings and sub-catchment discretization levels. Parajuli *et al.* (2008), predicted approximately 73% of sediments reduction with a Vegetated Filter Strip (VFS) of 10 m width. They also concluded that a 15 m and 20 m VFS removed up to 82% and 89% of sediments yield. The results of Parajuli *et al.* (2008) also affirms that the impact of filter strip on sediments yield is influenced by their respective width.

In Ethiopia and Eritrea, performance of selected management measures have been studied (Herweg and Ludi, 1999). Herweg and Ludi (1999) demonstrated that filter strips reduced soil loss by 81%, 60%, 84% in Afdeyu, Hunde Lafto and Gununo, respectively. Studies reveal that implementing soil and water conservation measures such as filter strips would improve the functioning of the previously degraded ecosystems (Geertsma *et al.,* 2011; Knoop *et al.,* 2012; Sand *et al.,* 2014).

2.4 Structural Conservation Measures

Structural conservation methods are measures that involve putting up physical structures to control runoff and erosion.

2.4.1 Terraces

Terraces are earthen embankment constructed in the dominant slope partitioning the field in uniform and parallel segments (Morgan, 2005). The American Society of Agricultural Engineers (ASAE) categorized terraces based on cross-section, alignment, grade and outlet thus four types of terraces are found. They include; broad based, narrow-based, bench and steep backslope terraces.

Fanya juu terraces have been implemented in Kenya as a modified form of bench terraces (Kiome and Stocking, 1995; Morgan, 2005; Saiz *et al.,* 2016). In approximate 7 years, a fanya juu terrace can develop into a level terrace due to deposition of sediments along the slope (Herweg and Ludi, 1999). Natural terraces can be formed through the use of vegetative measures (Angima *et al.,* 2003). Angima *et al.* (2003) reported that the use of napier grass in developing and stabilizing terraces is a good practice that enhance sediment reduction, fertilizers and pesticides. Terraces have been demonstrated to reduce overland flow and soil erosion in a SWAT modelling of soil and water conservation methods in the Upper Blue Nile Basin of Ethiopia (Lemann *et al.,* 2016).

According to Bracmort *et al.* (2006), terraces are installed to reduce mainly sheet and rill erosion from agricultural areas. Bracmort *et al.* (2006), modelled the impact of structural conservation methods on sediment and phosphorous yield. They concluded that terraces and field borders can be used to reduce sediment and phosphorus from non-channel-upland areas in Black Creek sub-catchments. Elsewhere in Tanzania, terraces have been used to improve soil composition, moisture level and increase yields due to reduced erosion and improved soil fertility in Ulugurus mountains (Branca *et al.,* 2011).

Reduction of sediment and erosion from the fields means that ecosystems services are enhanced and the cost associated with the unsustainable use of the ecosystems is reduced e.g. the cost of treating raw water. Terraces reduces high amount of nutrient and sediments from surface runoff compared to other management practices (Arabi *et al.,* 2008). However, their establishment can be very costly for small scale farmers. Inadequate capital can result in sub-standard terraces that may actually increase soil erosion when runoff accumulates at low points (CARDI, 2010).

Land degradation by mass movement of the soil can be increased when more water is retained in situ (CARDI, 2010). Crops yields may decline in places where the top soil is removed and crops are grown on the less fertile subsoil. Studies have shown that not all slopes are suitable for terracing. Slopes less than 2.3% would actually result to more soil erosion or mass movement of soils while slopes gradient greater than 10% implies that the spacing between terraces would have to be increased (Arabi *et al.,* 2008).

Establishing fanya juu terraces in Kenya on a slope of 5-8% would cost approximately US$ 320 per hectare (WOCAT, 2007). Additionally, the cost of maintaining the terraces is approximated as US$ 38 per hectare. Although this is cheaper compared to other types of terraces (WOCAT, 2007), the cost of implementing them would be expensive for small scale farmers and lack of investment funds would limit their adoption. Recommendations of a study conducted in China indicate that terraces should only be implemented when other agronomic means have proven to be ineffective or not feasible (Shi *et al.,* 2004). However, in Kenya, terraces are widely encouraged in arid and semi-arid areas due to their water conservation potential and increased crop productivity (Kiome and Stocking, 1995).

2.4.2 Grassed waterways

According to Humberto and Rattan (2008), grassed waterways are channels of water having grass established along drainage pathways so as to reduce surface runoff and sediments. The vegetation planted along the waterways reduce the velocity of surface runoff thereby allowing the sediment to settle down. Grassed waterways are implemented in areas where runoff concentrates to convey it to the outlet (Fiener and Auerswald, 2006a). Unlike filter strips, grassed waterways are wider and longer and they are installed in the thalweg (drainage ways) (Evrard *et al.,* 2007).

Terrain in a grassed waterway can be divided into two sections where the shallow sheet flow enter the grassed waterway at the side slopes from the adjacent fields (Fiener and Auerswald, 2005). The second section is the area of flow concentration (thalweg) along the base of channel. This section could also be the conveyance channel of water from a terrace system. The efficiency of the latter section mainly depends on the velocity of the surface runoff, the vegetation type and characteristics, the length and width of the grassed waterway (Fiener and Auerswald, 2006). Grassed waterways with large hydraulic roughness portrays high reduction in velocity of the surface runoff and sediment yield from the uplands (Fiener and Auerswald, 2006b).

Grassed waterways have been studied as an effective way of reducing sediments and pollutants from the agricultural lands and urban areas (Evrard *et al.,* 2007). The roughness of the channel defined by Manning's coefficient, channel erodibility and channel cover factor are some of the parameters that influence the effectiveness of the grassed waterways (Bracmort *et al.,* 2006). Research has shown that grassed waterways can have high reduction in sediments and other contaminants transported in surface runoff (Fiener and Auerswald, 2006). In a laboratory experiment to determine the effectiveness of grassed waterways Briggs *et al.* (1999) found that grassed water ways to reduced runoff amount by 47% while herbicides were reduced by 56% compared to non-grassed waterways. Tuppad *et al.* (2010) simulated management alternatives for catchment protection and reported that grassed waterways reduced sediment yield by 44% initial simulation value.

A combination of two or more soil conservation methods can be used to improve the ecosystem services. Soil erosion and sedimentation hazards may be controlled in over 90% of the agricultural fields if combined conservation methods are employed in a catchment (Shi *et al.,* 2004). This study sought to evaluate the effectiveness of selected conservation methods in reducing sediment yield and also assess their effectiveness when they are combined.

CHAPTER 3
MATERIALS AND METHODS

3.1 Study Area

3.1.1 Location and hydrological characteristics

Thika-Chania catchment spans three counties in Kenya i.e. Kiambu, Murang'a and Nyandarua as shown in Figure 3.1. The catchment lies between latitude 36.58º and 37.58º E and 0.58º and 1.17º S (Adamtey *et al.,* 2016; Kigira *et al.*, 2010). Thika and Chania rivers are the main drainage systems in the catchment. At the outlet of the catchment, the two rivers join and then drains into Tana River to Masinga reservoir. Ndakaini and Sasumua reservoirs that are important source of water supplied in Nairobi are found in the catchment at (0.82 S, 36.850 E) and (0.75 S, 36.667 E), respectively.

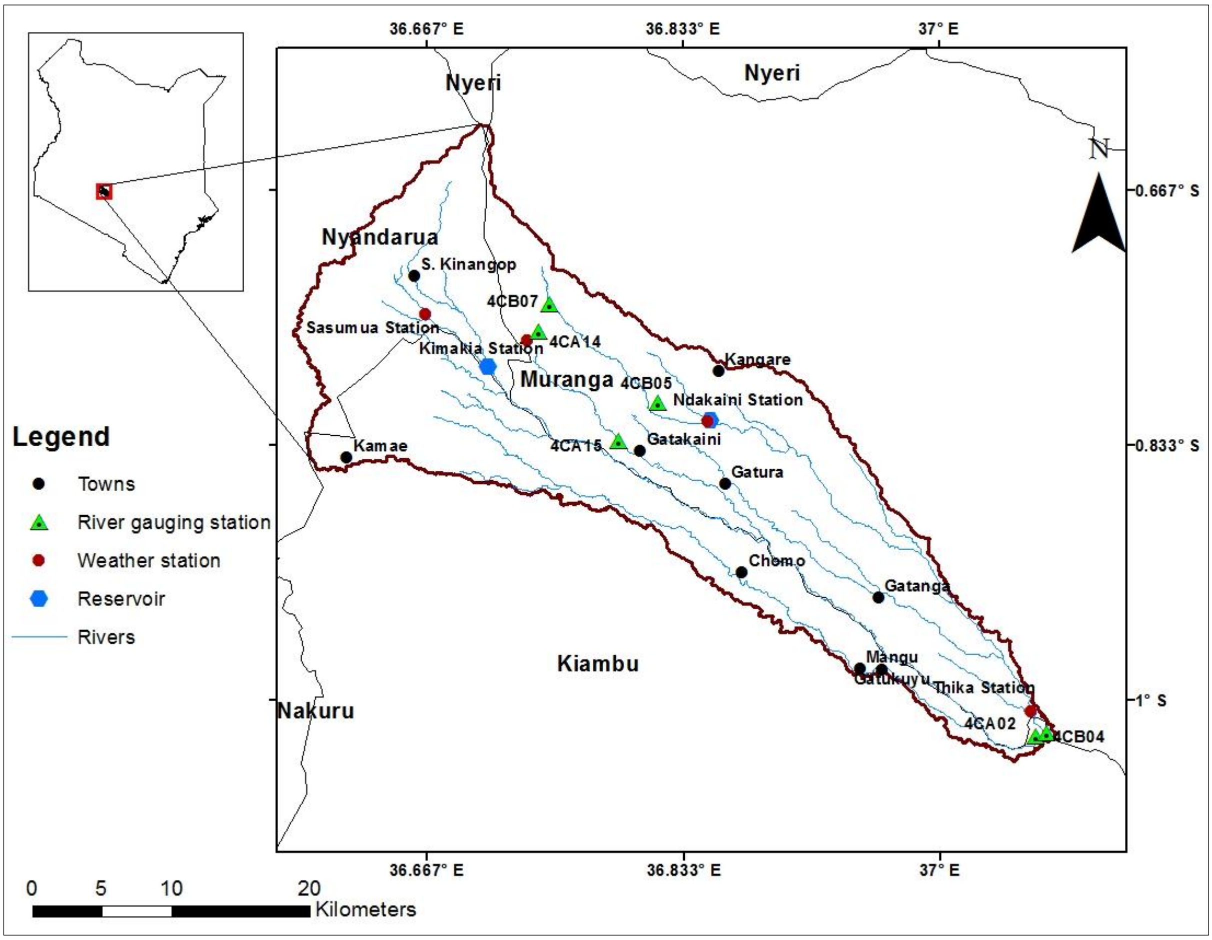

Figure 3.1 Location of Thika-Chania catchment

3.1.2 Topography

The elevation of the catchment can be visualized from the DEM of the catchment in Figure 3.2. The altitude varies from the source of Thika river at a maximum elevation of 3861 m above sea level (a.s.l) and the confluence of Thika and Chania river minimum elevation of 1449 m. a.s.l in the catchment.

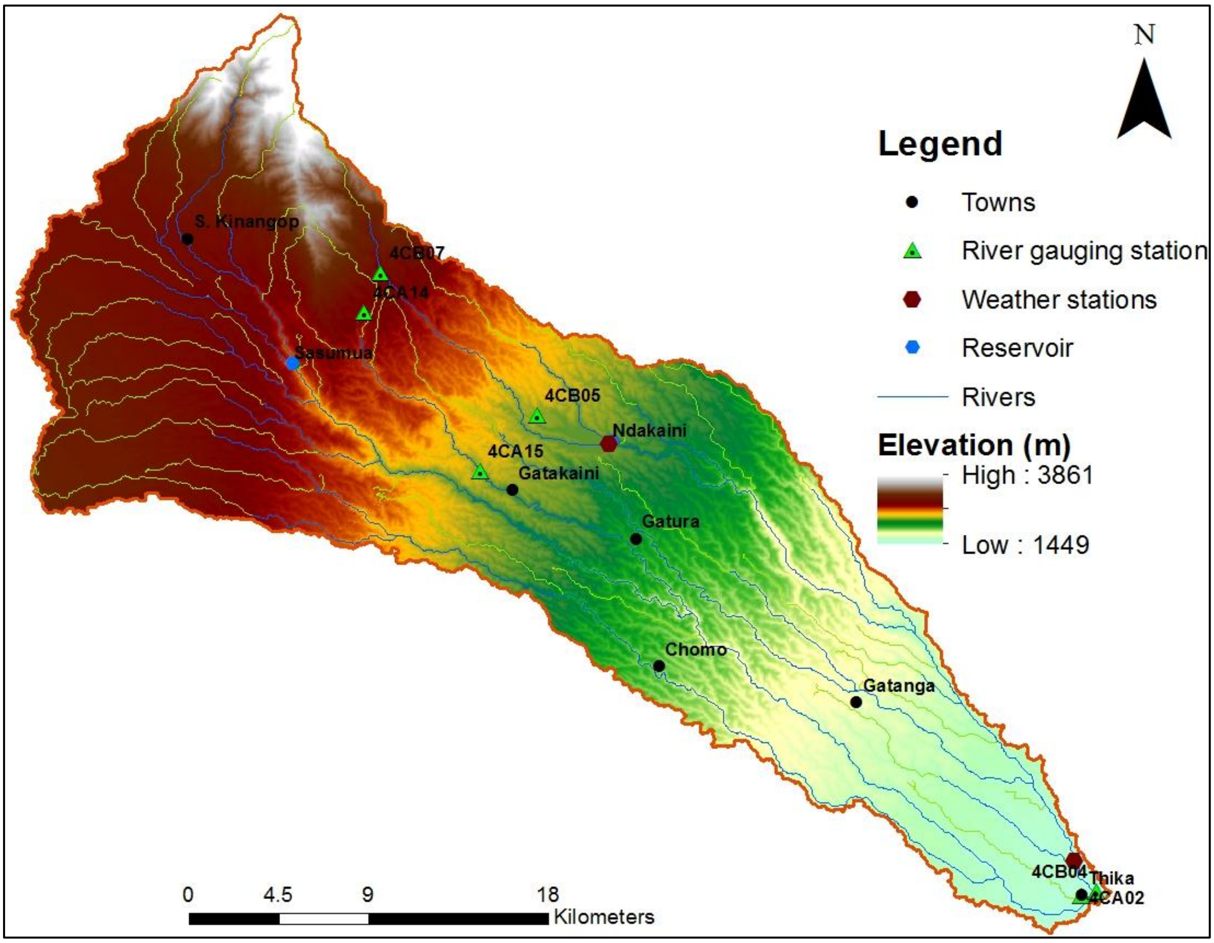

Figure 3.2 Thika-Chania catchment DEM

The predominant slopes in Thika-Chania catchment are between 0 and 20% (Figure 3.3) accounting for about 70% of the total area. Approximately 30% of the total catchment area has slopes in the range of 0-10% while slopes between 10-20% are found in 43% of the catchment area. The slopes in the upper parts of the catchment are relatively flat and prone to soil erosion and flash floods (Mwangi *et al.*, 2015). In the middle parts of the catchment the slopes are steep and have high erosion rates (Hunink *et al.*, 2013) while towards the outlet of the catchment the slope is comparatively flat.

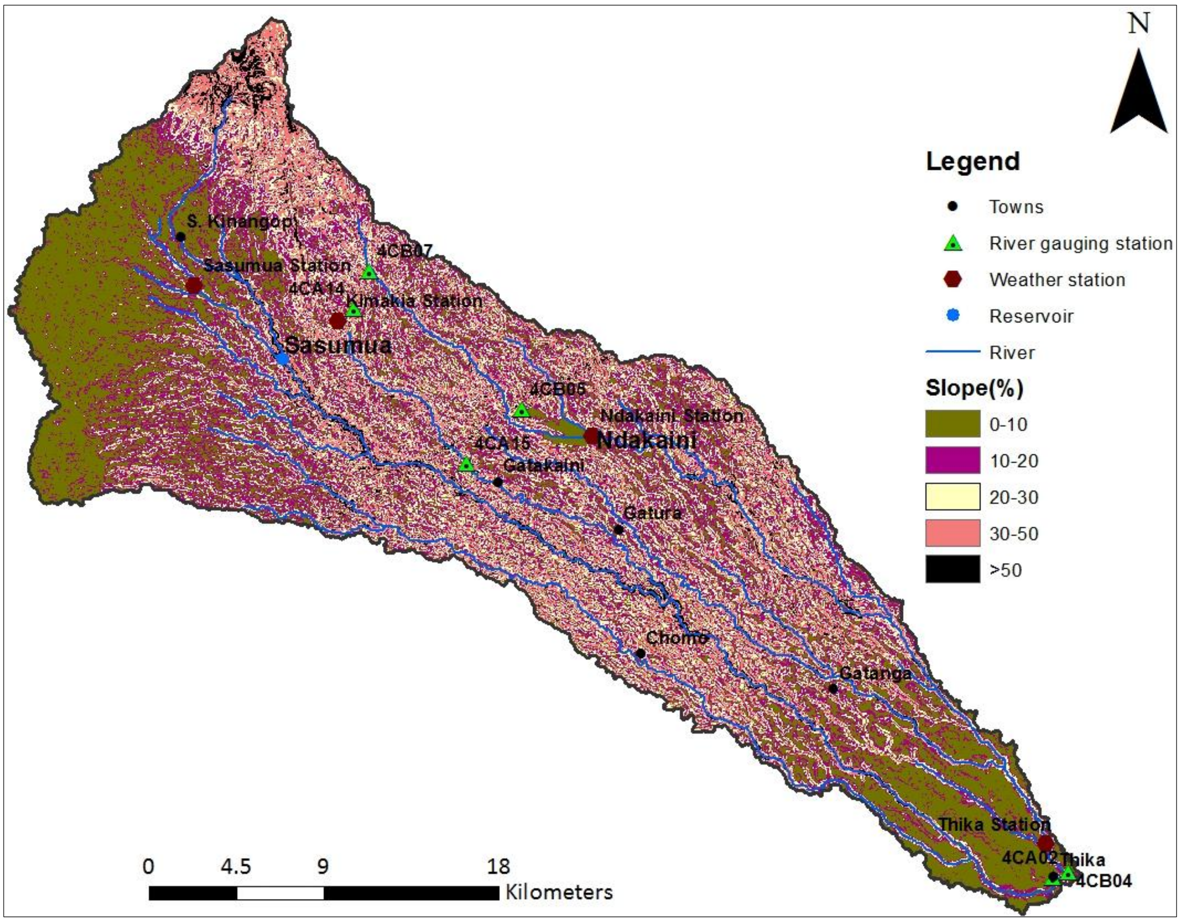

Figure 3.3 Slope variation in the catchment.

If slopes are considered as a factor influencing soil erosion, the middle region of the catchment is prone to soil erosion based on it higher slopes compared to other regions of the catchment. The agricultural lands have slopes ranging from 0-30 % while the steeper slopes are found in forests and tea zones. The region also has most of its land use on agriculture just after the forested region (Figure 3.4) and hence the need to have soil conservation measures employed in the region.

3.1.3 Land use

The major land use in Thika catchment include rain-fed agriculture, forestry and shrub land (Hunink *et al.,* 2013). Dense agricultural activities are practiced in the upper and middle parts of the catchment where tea, maize and coffee are mainly grown. Agricultural practices in the catchment is mainly rainfed and extensive cultivation in previously uncultivated areas has been observed (Vogl *et al.*, 2017). A wide forest cover is found in the upper parts of the catchment. The forest separates the agricultural lands and help in maintaining water quantity and quality (Vogl *et al.,* 2017). The forested areas cover 35.6% of the total catchment area while 28.1%

and 19.6% of the area is on coffee and gen. agriculture, respectively. The areas classified as gen agriculture include those areas on horticultural products like vegetables, potatoes and other horticultural crops cultivated in small scale.

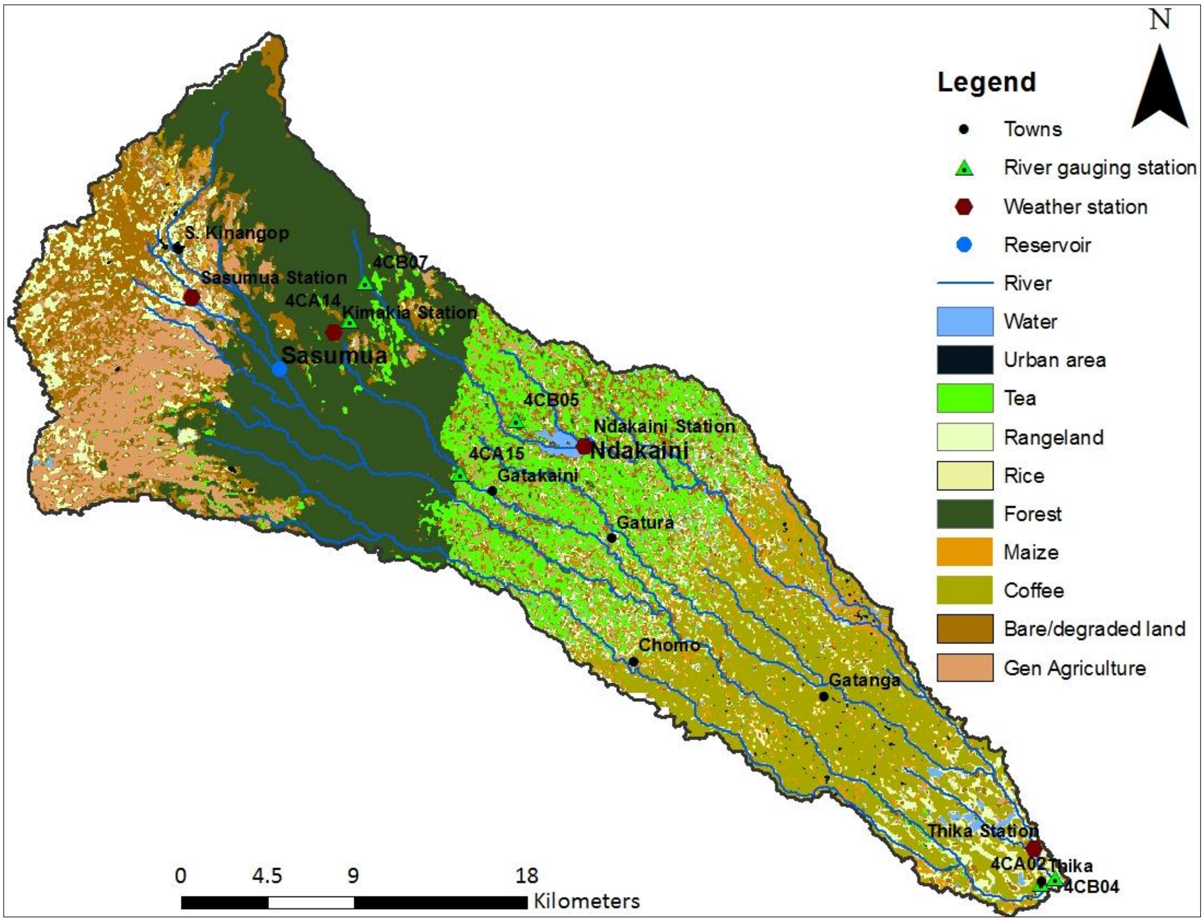

Figure 3.4 Land use in Thika-Chania catchment (Wilschut, 2010)

3.1.4 Climate

Rainfall in the catchment varies from low to high altitude areas in the catchment. Distribution of rainfall is bimodal with high peaks beginning from March and May and short rains coming in October to December as shown in Figure 3.5.

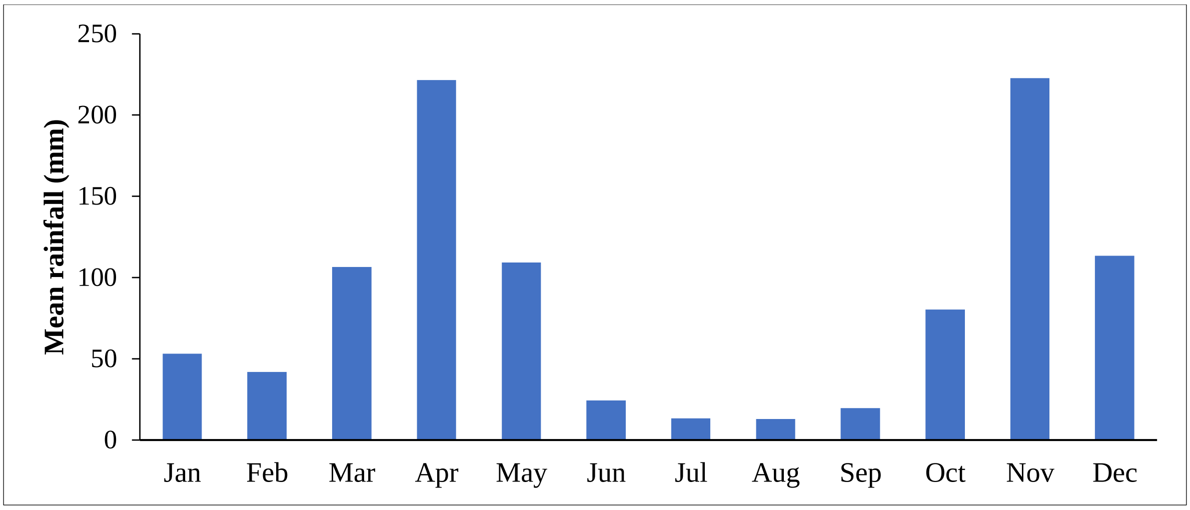

Figure 3.5 Bimodal rainfall distribution in Ndakaini station

The rainfall varies from 800 mm in low altitude areas to 2200 mm in high altitude areas of the catchment (Jaetzold and Schmidt, 2006; Mwangi *et al.*, 2015; Vogl *et al.*, 2016). High temperatures are experienced in low altitude areas ranging from 25º C to 30º C whereas in high altitude the temperatures range from 18º C and 20º C (Adamtey *et al.*, 2016). The average potential evapotranspiration rate in the catchment is approximately 1200 mm annually (Jaetzold and Schmidt, 2006).

3.1.5 Soils

The dominant soils in the catchment are Umbric Andosol and Humic Nitisols as shown in Figure 3.6. The physical and chemical properties vary spatially and also between layers. The soil erodibility factor of the soils in the catchment varies from 0.07 to 0.21 (Angima *et al.*, 2003; Hunink and Droogers, 2011). From Figure 3.6, Thika-Chania catchment has more than 10 soil types that supports different land uses and with different physical and chemical properties. Umbric Andosol is the dominant soil type in forested areas of the upper parts of the catchment. Umbric Andosol, Humic and Rhodic Nitisol are predominantly found in the middle and lower parts of the catchment where agriculture is main land use (Adamtey *et al.*, 2016).

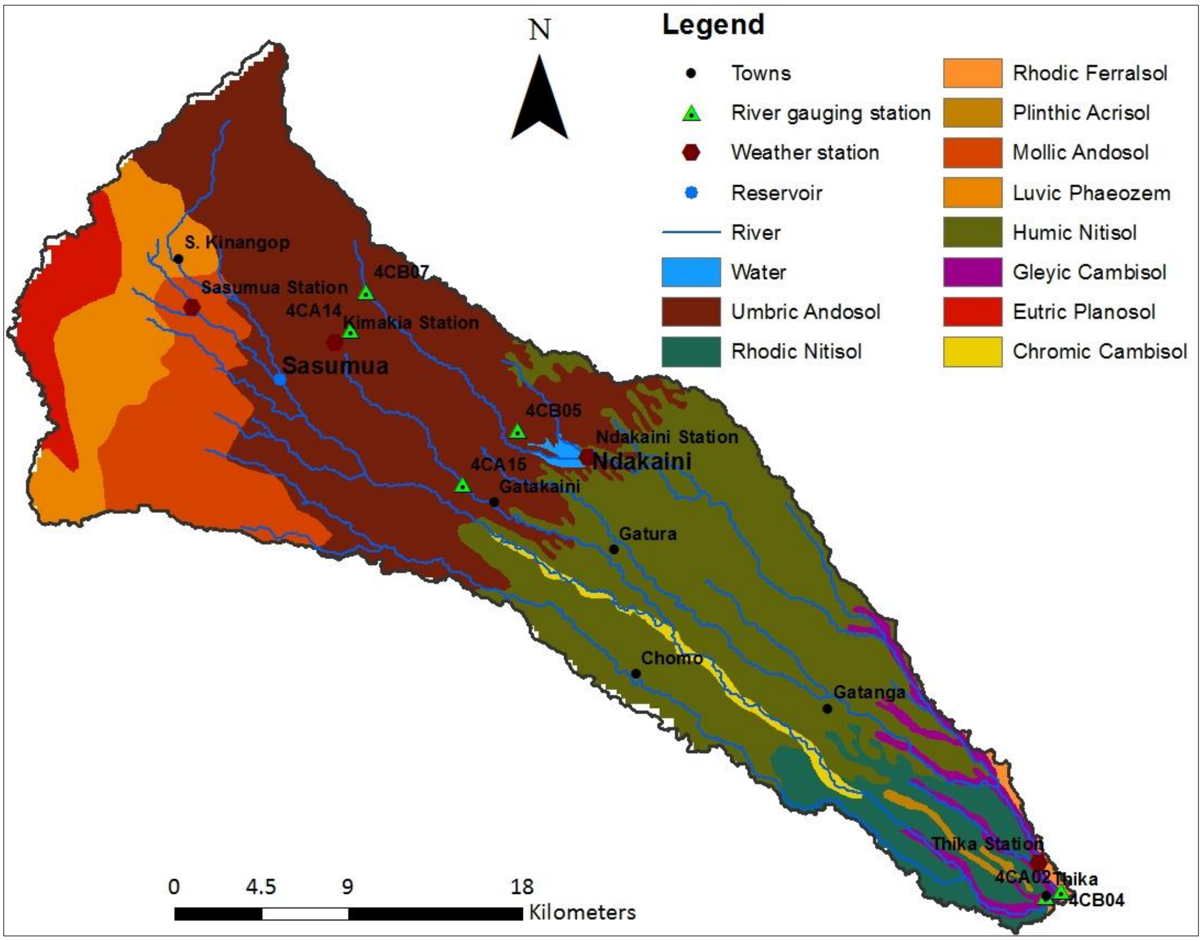

Figure 3.6 Soils map for Thika-Chania catchment

The low-lying areas of the catchment constitutes the Gleyic Cambisol and Rhodic Nitisol (NAAIAP, 2014). Soil parameters required for the modelling of the ecosystem services in this study include; saturated hydraulic conductivity, maximum rooting depth, bulk density, soil erodibility factor, percent organic carbon, available moisture content, soil textural composition, soil pH, Albedo and the hydrologic group of the soil.

3.2 Data Acquisition, Calibration and Validation of the SWAT Model

3.2.1 DEM, land use and soil data collection

A Shuttle Radar Topography Mission Digital Elevation Model (SRTM-DEM) with a 30-meters resolution was used to analyze the topography of the catchment including slopes, channel network and catchment delineation.

Spatial information on land cover was obtained from a raster land use map (2009) of Upper Tana that is based on remote sensing for land classification (Wilschut, 2010). The land use map was obtained from International Soil Reference and Information Center -World Soil

Information (ISCRIC-WISE) as a raster data with 30-meters resolution that provides a more accurate description of the variation of the land use types in each zone of the catchment. The map resolution corresponds with the resolution of the digital elevation model hence there was no need for further analyses on their resolutions.

A digital soil map of the Upper Tana River was downloaded from the ISRIC – World Soil Information (http://geonode.isric.org/layers/geonode:sotwis_ket_soil_unit_composition) on 13th April, 2017. The map was derived from the Soil and Terrain for the Upper Tana (SOTER_UT with a scale of 1:250,000 and the ISRIC-WISE soil profile database. The harmonization of the soil properties was achieved using a standardized taxonomy-based pedo-transfer process in order to come up with a SOTER-based soil parameter estimates (SOTWIS) for the Upper Tana catchment in which Thika catchment is located (Batjes, 2011). The information derived from this map include the soil texture, available water content, hydraulic conductivity, organic carbon content and bulk density. The corresponding soil properties for each of the soil type in the catchment was obtained from soils parameter database (SOTWIS).

3.2.2 Stream flow and sedimentation data

Data on stream flow and inflow to Ndakaini dam were collected from the Water Resources Management Authority (WRMA) and Nairobi City Water and Sewerage Company (NCWSC). The data was then checked for consistency and accuracy using Microsoft excel program. Streamflow data gaps were filled using simple linear interpolation from the neighboring gauging stations with approximate flow rates and trends (Begou *et al.*, 2016). Sediments concentration data obtained from bathymetric survey conducted in 2011 was used to set up the model for the base scenario (Hunink and Droogers, 2011). The bathymetric survey also contains information on sedimentation of Sasumua and Ndakaini dam. The stream flow data between 1980 and 2010 was used to calibrate and validate the model due to the continuity of the dataset. Figure 3.7 shows the stream flow variation in gauge 4CB05 that was used in the calibration process of the model.

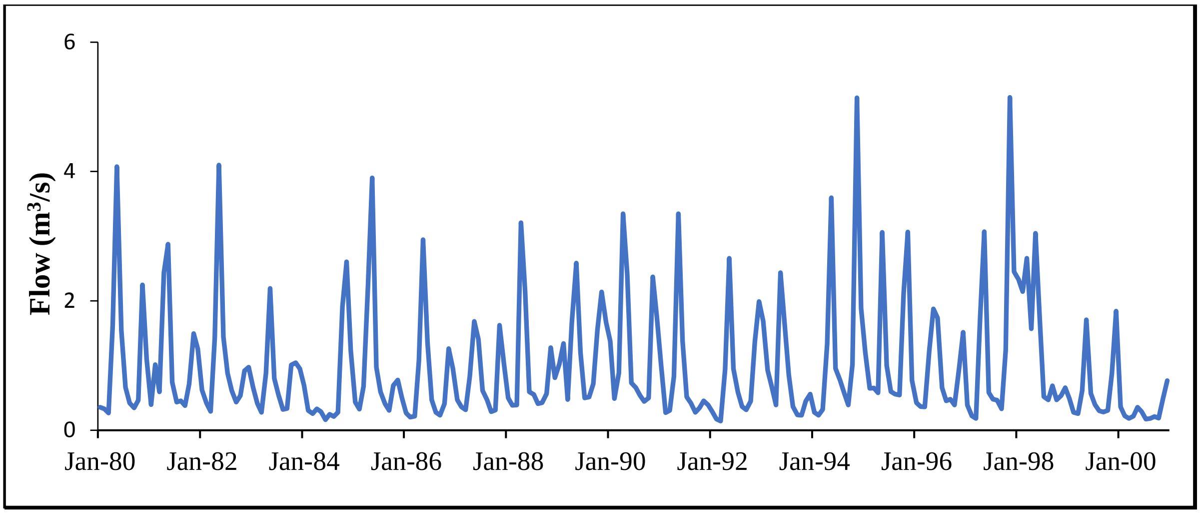

Figure 3.7 Streamflow for gauge 4CB05

To check the model for spatial accuracy in simulation of stream flow, the model was validated using data observed in a different gauging station (4CB04) in the catchment (Arnold *et al.*, 2012). Other studies conducted in Africa have used both the split sample and split-location approach to calibrate and validate the SWAT model for stream flow (Begou *et al.*, 2016).

3.2.3 Climatic data

Daily climatic data that includes rainfall, maximum and minimum temperature, solar radiation, relative humidity and wind-speed were obtained from various weather stations in the catchment. These stations include; Kimakia station, Sasumua dam station, Thika Dam and Thika agrometeorological station. In SWAT model, the daily weather data can be input from the observed dataset or simulated using weather generator (WXGEN) where data from neighboring stations are used (Arnold *et al.*, 2012). The model uses the climatic data of the station that is closest to the centroid of each sub-catchment (Abbaspour *et al.*, 2007).

Climatic datasets were checked for accuracy and consistency. The few missing data were filled based on correlation with data from other stations (Begou *et al.*, 2016). It was noted that some rainfall values were unrealistically large, for instance a daily rainfall of 684 mm. These data were deleted and replaced with an average value from the neighboring stations. Gaps in the climatic data were filled using statistical methods and linear interpolation from neighboring stations.

3.2.4 Model setup

The datasets used in setting up the model are described in Table 3.1.

Table 3.1 *Data used in SWAT model set up*

Datasets	Detail	Sources
Digital elevation model	30 m resolution	SRTM
SOTER-UT soils map	Scale 1:250000	ISRIC-WISE
Upper Tana Land use map (2009)	30 m resolution	ISRIC-GWC
Meteorological data	Daily (1996-2013)	Kenya Meteorological Department, NCWSC
Stream flow data	Daily (1996-2013)	WRMA, NCWSC
Sediments load/Turbidity data	Point data 2010	WRMA, NCWSC
Bathymetric survey data	Thika dam, Sasumua dam, Masinga dam	Bathymetric survey (2011)

The digital elevation model, soil map and land use maps were projected to the same coordinate system. After data preparation for upload into the model, the catchment was delineated automatically using the SWAT interface. The delineation process consisted of uploading the DEM, definition of the stream network and establishing the inlet and outlet of the catchment. The Critical Source Area (CSA) that controls the number of sub-catchments and channel network in the catchment was then determined. The bigger the CSA, the smaller the number of the sub-catchments and channel network density. The CSA therefore determines the minimum area that is required for initiation of channel flow. A user defined critical source area of 1200 ha was adopted since the average sub-catchment area fell within the range of 2-5% of the total catchment area (Jha *et al.*, 2004).

The projected land use map was then uploaded into the model and reclassified to match the SWAT land-use classes. To reclassify the land use type, a user look-up table was prior made as to identify the SWAT land use codes that corresponds to the land use classes obtained from the map. The soils shapefile was also uploaded and was automatically converted to a grid with a cell size equal to the DEM of the catchment. In case the overlap would not be satisfactory between the soils dataset and the catchments, an error message would be reflected (Winchell *et al.*, 2013). In this research, the overlap was 99.19% which was considered appropriate.

Key parameters missing in the SOTER soil database is the saturated hydraulic conductivity (K_S) and USLE soil erodibility factor (K_{USLE}) for each soil type and layers. The saturated hydraulic conductivity was estimated using the method of Jabro (1992) illustrated in equation (3.1);

$$\log(K_s) = 9.56 - 0.81\log(\%silt) - 1.09\log(\%clay) - 4.64(BD) \qquad (3.1)$$

where K_s is the saturated hydraulic conductivity (cm/h) and BD is the soil bulk density (g/cm^3). This method has also been used in other studies such as Hunink and Droogers (2015) and Maeda *et al.* (2010) to determine the saturated hydraulic conductivity.

The USLE erodibility factor (K_{USLE}) was estimated using the method of Williams (1995) as given in Neitsch *et al.* (2005).

$$K_{USLE} = f_{csand} \times f_{cl-si} \times f_{orgc} \times f_{hisand} \qquad (3.2)$$

where, f_{csand} is a factor that gives low soil erodibility for soils with high coarse-sand contents and high values for soils with little sand content, f_{cl-si} reduces the erodibility for soil with a high clay to silt ratio, f_{orgc} reduces the erodibility for soils with high organic carbon content and f_{hisand} reduces soil erodibility for soils with extremely high sand contents.

$$f_{csand} = \left(0.2 + 0.3 \times exp\left[-0.256 \times m_s \times \left(1 - \frac{m_{silt}}{100}\right)\right]\right) \qquad (3.3)$$

$$f_{cl-si} = \left(\frac{m_{silt}}{m_c + m_{silt}}\right)^{0.3} \qquad (3.4)$$

$$f_{orgc} = \left(1 - \frac{0.25 \times orgC}{orgC + exp[3.72 - 2.95 \times orgC]}\right) \qquad (3.5)$$

$$f_{hisand} = \left(1 - \frac{0.7 \times \left(1 - \frac{m_s}{100}\right)}{\left(1 - \frac{m_s}{100}\right) + exp\left[-5.51 + 22.9 \times \left(1 - \frac{m_s}{100}\right)\right]}\right) \qquad (3.6)$$

Where m_s is percent sand content, m_{silt} is the percent silt content, m_c is the percent clay content and orgC is the percent organic carbon of the layer.

The SWAT user soils database was modified to include the soils of the catchment as shown in Figure 3.8. A user look-up table was made for the reclassification of the soils. The soil map was then reclassified to correspond with the SWAT user soil database.

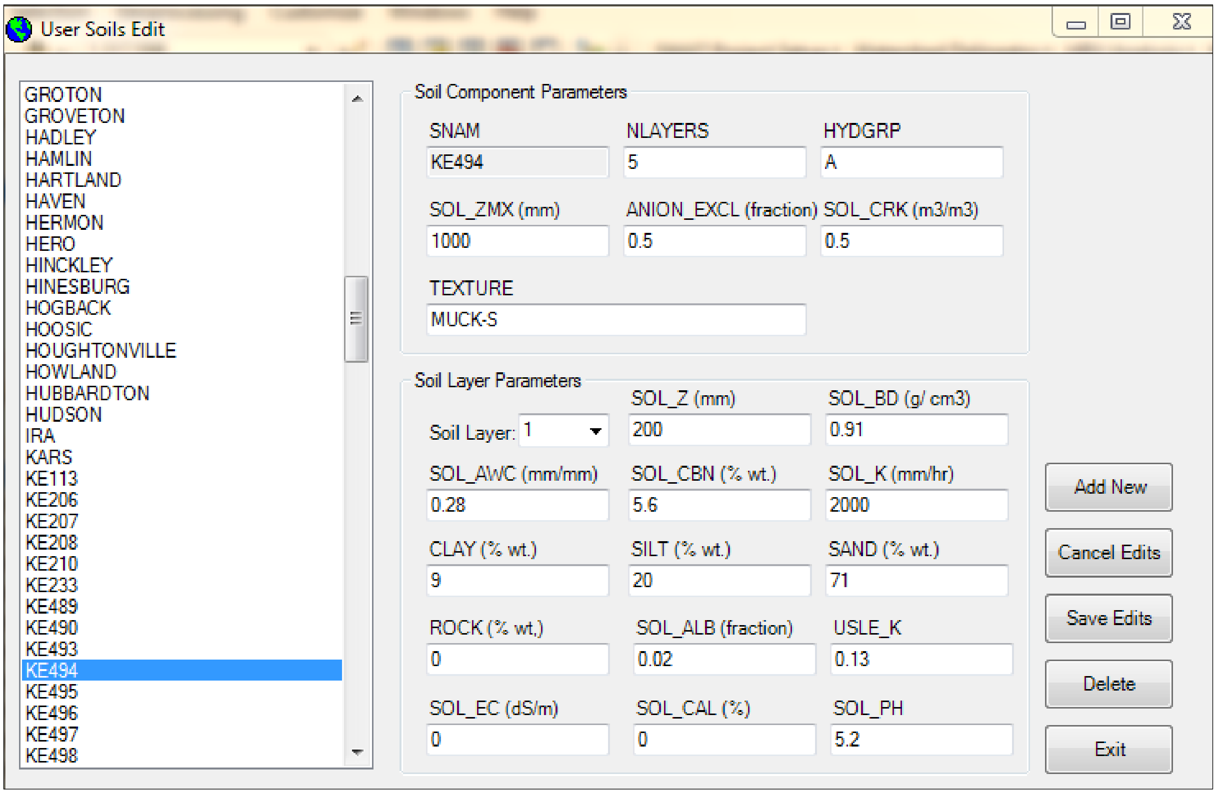

Figure 3.8 User soils editing interface for SWAT

The Hydrologic Response Unit (HRU) which represents an area of similar land use, soils and slopes use was then established. Multiple HRUs were used in this study where a threshold of 30% for land use, 20% for soils and 30% for slopes was adopted. This means that if a 20% threshold for land use is used, only land use that is more than 20% in a given area would be considered in HRU distribution. In that regard, land use, soils and slopes that did not meet the minimum percentage for HRU distribution were eliminated. The land use, soils and slopes were then reallocated such that 100 percent of the sub-catchment area is included in the simulation.

The weather input parameters obtained from the meteorological stations were uploaded to the model. SWAT requires daily time steps of rainfall, maximum and minimum temperature, humidity and windspeed. The SCS curve number was used to compute the surface runoff and flow routing through the stream utilized the Muskingum routing method. To determine the evapotranspiration rate, the Penman Monteith formula was applied in this study.

3.2.5 Model parameter sensitivity analysis

Main challenges in setting up a model is the scarcity of various levels of temporal and spatial data. As such, the calibration and validation of erosion model is challenging due to the dynamic and non-linear responses that characterize basin-scale sediment transport and model or process uncertainties (Abbaspour *et al.,* 2007; Mwangi *et al.,* 2015). Parameter sensitivity analysis was conducted to guide the calibration process and to identify parameters that are most sensitive and have the great impact on the model outputs and also the precision parameters required for calibration (Arnold *et al.,* 2012). The Sequential Uncertainty Fitting version 2 in SWAT Calibration and Uncertainty Programs (SWATCUP SUFI2), was used to analyze global sensitivity of the parameters. After one run of the Latin hypercube sampling, the initial uncertainty ranges were assigned to the respective parameters. Large parameters uncertainties were initially assumed so as to ensure that the observed data would be captured within the 95 Percent Prediction Uncertainty (95 PPU) (Abbaspour *et al.,* 2007). The parameters ranges were then adjusted after every program run until the 95 PPU band brackets most of the observed data. Data of low quality, for instance having many outliers, may have less than 0.5 of the data bracketed within 95 PPU (Abbaspour, 2015). For the present study, more than 50% of data bracketed within then the 95 PPU was considered satisfactory. The objective functions were then evaluated depending on the defined thresholds (Abbaspour, 2015).

3.2.6 Calibration and validation of the model

Stream flow data for gauging station 4CB05 between 1998-2013 was used in calibration and validation process to reduce the prediction uncertainty of the model. This period was chosen as it provided a convenient time range for calibration and validation where the data is not prone to errors and gaps and also fell within the period the land use map was developed. The data was then split into two; one set used for calibration and the other used in validation of the model. The first two years in the calibration process were set as the warm up period for the model.

Model parameters set during calibration process were left constant during the validation process. The simulated flows were compared with the observed flows through statistical methods and visual examination of the graphical results. Three statistsical performace indicators of the model namely; coefficient of determination (R^2), Percent BIAS (PBIAS) and

Nash-Sutcliffe coefficient (NSE) were used (Abbaspour, 2015; Moriasi *et al.*, 2007; Nash and Sutcliffe, 1970). These indicators are calculated using equations (3.7), (3.8), and (3.9);

$$R^2 = \frac{\left[\sum_i (Q_{m,i} - \bar{Q}_m)(Q_{s,j} - \bar{Q}_s)\right]^2}{\sum_i (Q_{m,i} - \bar{Q}_m)^2 \sum_i (Q_{s,j} - \bar{Q}_s)^2} \quad (3.7)$$

$$NSE = 1 - \frac{\sum_i (Q_m - Q_s)^2}{\sum_i (Q_{m,i} - \bar{Q}_m)^2} \quad (3.8)$$

$$PBIAS = \frac{100 \times \sum_{i=1}^{n} (Q_m - Q_s)_i}{\sum_{i=1}^{n} (Q_{m,i})} \quad (3.9)$$

Where Q_m is the measured discharge, Qs is the simulated discharge, $\bar{Q}_m$is the average measured discharge and $\bar{Q}_s$ is the average simulated discharge.

The less the value of PBIAS, the better the simulation results of the model. The optimum value of the PBIAS is 0 (Arnold *et al.*, 2012). Positive values indicate underestimation whereas negative values indicate over estimation. The values of the coefficient of determination can range from 0 to 1. A value of one indicates a perfect correlation between the observed and simulated values (Arnold *et al.,* 2012). The coefficient of determination (R^2) shows the closeness of the observed values to the simulated values. In this study, values of R^2 greater than 0.5 were considered satisfactory. Similarly, the Nash-Sutcliffe coefficient (NSE) was used to check whether the simulated data corresponds to the observed data. NSE values ranges between -∞ to 1. A value of 1 indicates a perfect match between the observed and simulated data. To evaluate whether the NSE and PBIAS values are satisfactory, the guidelines provided by Moriasi *et al. (*2007) were used.

The model was validated using data from 2006-2013 for gauging station 4CB05 without changing the calibration parameters. To check whether the model correctly simulated the observed data spatially, graphical method was used to compare the observed and simulated values. for gauging station 4CB04 between the years 1998-2008. Begou *et al.* (2016) applied a similar methodology during streamflow calibration for Bani catchment in Mali.

Sediment data from bathymetric survey conducted in 2011 was used to manually calibrate and validate the model. This was achieved by adjusting the MUSLE parameters to match the observed values within and at the catchment outlet. The model was validated for sediments based on the methodology used by Mwangi *et al.* (2015) and Vogl *et al.* (2016) in upper Tana.

The calibrated and spatially validated model was then run to form the base scenario for simulating impacts of agronomic, vegetative and structural conservation measures on sediment and water yield.

3.3 Simulating Agronomic, Vegetative and Structural Conservation Measures

The methodology applied for simulation of conservation methods was adopted from a study by Arabi *et al.* (2008). Process parameters that mimic the functionality of the conservation practices were identified and adjusted to represent a particular conservation effort.

3.3.1 Contour farming

Contour farming as a soil and water conservation method was assumed to be implemented on 50% of the catchment area. These include areas on general agriculture, coffee, maize and bare/degraded lands. Contour farming was simulated by varying the values of the SCS curve number (CONT_CN) and the USLE practice factor (CONT_P). CONT_CN influences the surface runoff and sediment yield from a given catchment. Decreasing the value of CONT_CN would therefore imply a reduction in surface runoff and hence sediment yield. The values of CONT_CN were reduced by 3 units from the base scenario value to simulate practicing contour farming (Arabi *et al.*, 2008). The USLE practice factor was also adjusted from the base value of the respective slope class of each HRU as shown in Table 3.2. The percent reduction of surface runoff and sediment yield was computed and a map produced to show the areas where contour farming would be most effective.

Table 3.2 USLE practice factor (CONT_P) for contour farming

Land slope (%)	CONT_P
1 – 2	0.60
3 – 5	0.50
6 – 8	0.50
9 – 12	0.60
13 – 16	0.70
17 – 20	0.80
21 – 25	0.90

Modified from Arabi *et al.* (2008); Wischmeir and Smith (1978)

3.3.2 Vegetative filter strips

There are two methods of modeling filter strips in SWAT; the trapping efficiency method and the area ratio method (Ha and Wu, 2015). According to Waidler *et al.* (2009), the trapping efficiency method ($Trap_{eff\text{-}sed}$) uses the strip width of the filter (FILTERW) while the latter uses the ratio of field area to filter strip area (VFSRATIO), fraction of the HRU which drains to the most concentrated 10% of the filters strip area (VFSCON), fraction of the flow within the most concentrated 10% of the filter strip which is fully channelized (VFSCH). Ha and Wu (2015) reported that the two methods simulate sediment reduction in a similar manner. The trapping efficiency method was adopted because it has been widely applied in other studies by Arabi *et al.* (2008), Bracmort *et al.* (2006) and Mwangi *et al.* (2015) to model filter strips.

It was assumed that the filter strip would be implemented in all agricultural lands of the sub-catchments, bare lands and along riparian areas. In SWAT, the width of the strip parameter (*FILTERW*) was adjusted to simulate its effect in trapping sediments. The filter strip was varied in width from 0 to a maximum of 45 meters at an increment of 5m in every simulation. The use of the filter strips in the reduction of the amount of sediments downstream depends on the trapping efficiency of the cover crop which is modelled by equation (3.10) (Arabi *et al.*, 2008);

$$Trap_{eff-sed} = 0.367 \times FILTERW^{0.2967} \tag{3.10}$$

Sediments yield from each run of the model at different filter width was recorded. The results of sediment yield against the filter widths was then plotted to determine the effective width for sediment removal by the vegetative filter strips. Although filter strips may be implemented along the farm/river edges, they do not influence the within-channel processes (Arabi *et al.*, 2008). Therefore, conservation methods that influence channel processes, for instance the grassed waterways, were evaluated independently.

3.3.3 Terraces

Terraces were assumed to be implemented in parallel across the fields based the methodology used by Arabi *et al.* (2008). It was assumed that reducing the slope length through terracing would reduce surface runoff volume and sediment yields. Therefore, to simulate the effects of terraces in conserving soil erosion and hence reducing sediment yield, the slope length (TERR_SL) in the HRUs with slopes greater than 2.3% was adjusted (Arabi *et al.*, 2008). The USLE-P factors (TERR_P) and the SCS curve number (TERR_CN) were also modified. To represent reduction in surface runoff the SCS curve number was reduced from the calibrated values as documented in Neitsch *et al.* (2005) and guided by the method of Arabi *et al.* (2008). The TERR_CN values were reduced by 6 units from the base scenario value. The recommended values of the TERR_P were obtained from literature as shown in Table 3.3 (Arabi *et al.,* 2008; Wischmeir and Smith, 1978);

Table 3.3 Terracing USLE P (TERR_P)

Slope (%)	TERR_P
1-2	0.12
3-5	0.1
6-8	0.1
9-12	0.12
13-16	0.14
17-20	0.16
21-25	0.18

Modified from Arabi *et al.* (2008)

The slope length indicated in SWAT as (TERR_SL) in terraces management option 1 (MGT_OP1) was modified based on equation (3.11) (Arabi *et al.,* 2008).

$$SLSUBBSN = (x \times SLOPE + y) \times \frac{100}{SLOPE} \tag{3.11}$$

Where x is a dimensionless variable with values from 0.12-0.24. A value of 0.24 is used for low rainfall areas and 0.12 used for high rainfall areas. Based on the climatic conditions in the catchment an average value of 0.18 was used. The dimensionless variable y is influenced by soil erodibility and crop management system. According to Arabi *et al.* (2008) the variable can take values of 0.3, 0.6, 0.9 or 1.2 as shown in Table 3.4.

Table 3.4 Typical values of the dimensionless variable y

Ground cover (%)	Soil erodibility factor		
	0-0.2	0.2-0.28	0.28-0.64
10	0.75	0.53	0.30
40	0.98	0.75	0.53
80	1.20	0.98	0.75

Source: *USDA NRCS, 2015*

A low y value of 0.3 is used for highly erodible soil with conventional tillage and little residue, while the high value of 1.2 is used for soil with very low erodibility and with a residue of more than 3.3 t/ha or no-till management condition (USDA NRCS, 2015). Thika-Chania catchment has soil erodibility values ranging from 0.07 to 0.24 depending on the soil types and cultivation practices (Angima *et al.,* 2003; Hunink and Droogers, 2011; Mati *et al.,* 2000). In agricultural areas, harvesting of crops leaves little or no crop residue on the lands hence an assumed ground cover of 40% was used. Based on this percentage and interpolation of soil erodibility factor in the catchment (Table 3.4), a y value of 0.75 was used in the current study. These parameters were adjusted simultaneously for every simulation

Approximately 50% of the catchment area was simulated for with terraces. These includes the areas on coffee, maize, bare/degraded land and general agriculture. Tea and forested areas were excluded because they were assumed to be characterized by good ground cover.

3.3.4 Grassed waterways

Implementation of grassed waterways were simulated in small seasonal channels and drain ways that convey water from agricultural lands and towns as given by Arabi *et al. (*2008) and Secchi *et al. (*2007). The method used in this study was based on the modelling option of grassed waterways in SWAT (Winchell *et al.,* 2013). Grassed waterways were modelled by modifying the scheduled management option (MGT_OP 7) in SWAT. Parameters used include GWATI that turns on/off the grassed waterway for simulation. Manning's roughness factor "n" (GWATn) for the main channel was adjusted based on recommendations of Arabi *et al.* (2008). A value of 0.1 was used to represent channel roughness and control the erosive energy of streams. The value also represents dense grasses under non-submerged conditions (Arabi *et al.,* 2008; Fiener and Auerswald, 2006b). Based on waterways installed along farmlands and road sections in the catchment (Mwangi *et al.,* 2015), a uniform channel width (GWATW) of 2.5 m and a depth (GWATD) of 0.3 m was assumed. Linear parameter for determining sediment re-entrained in the channel sediment routing (GWATSPCON) was taken as 0.005 (Waidler *et al.,* 2009). The average slope of the grassed waterway (GWATS) was determined based on the average slope of the main channel (Waidler *et al.,* 2009).

3.4 Simulating the Combination of Different Conservation Methods

Integration of different conservation measures was based on the widely practiced methods in the catchment. This information was obtained from baseline reviews, household surveys and reports conducted in the catchment and adjoining areas (Kyalo *et al.*, 2014; Leisher, 2013)

A combination of filter strips and contour farming was simulated to evaluate the effectiveness of the conservation measures when they are implemented simultaneously. Sediments and water yield were simulated when filter strips of different widths are implemented on a land where contour farming is practiced. This was achieved by adjusting the filter strip width parameter in SWAT, the USLE P and the SCS curve number. The results obtained on sediment yield were then compared to that when individual conservation methods are used.

To further determine the impacts of integrating terraces with filter strips, the HRU slope length, USLE P, filter strip width and the SCS curve number were adjusted to represent the two conservation methods being implemented on the same land. The use of an integration of soil and water conservation methods have been shown to improve crop yields and sediment reduction in surface runoff (Ouattara *et al.,* 2017).

Grassed waterways were combined with contour farming by adjusting the respective simulation parameters. These parameters include the SCS curve number, CONT-P, Manning roughness coefficient, width, slope and depth of the grassed waterway and GWATSPCON. Similarly, terraces and grassed water ways were also simulated by adjusting the slope length (TERR-L), TERR-P, TERR-CN and the parameters used to simulate grassed waterways.

The effectiveness of the conservation methods was evaluated by comparing the outputs of the model before and after their implementation using equation (3.12).

$$E = \frac{x_1 - x_2}{x_1} \times 100 \qquad (3.12)$$

Where E is the effectiveness of the conservation method, x_1 and x_2 are model simulation before and after implementation of the conservation method, respectively.

CHAPTER 4
RESULTS AND DISCUSSION

4.1 Model Parameter Sensitivity Analysis, Calibration, Validation and Initial Run

The average sub-catchment area was found to be 19.53 km^2 which represents approximately 2.3% of the total catchment area. These results met recommendations of Jha *et al.* (2004) that the average size of a sub catchment to effectively simulate ecosystem services should be between 2-5% of the total catchment area. The total catchment area after delineation was found to be 840 km^2. The entire catchment was subdivided into 43 sub basins as shown in

Figure 4.1 Thika-Chania sub-catchment, weather and river gauging stations

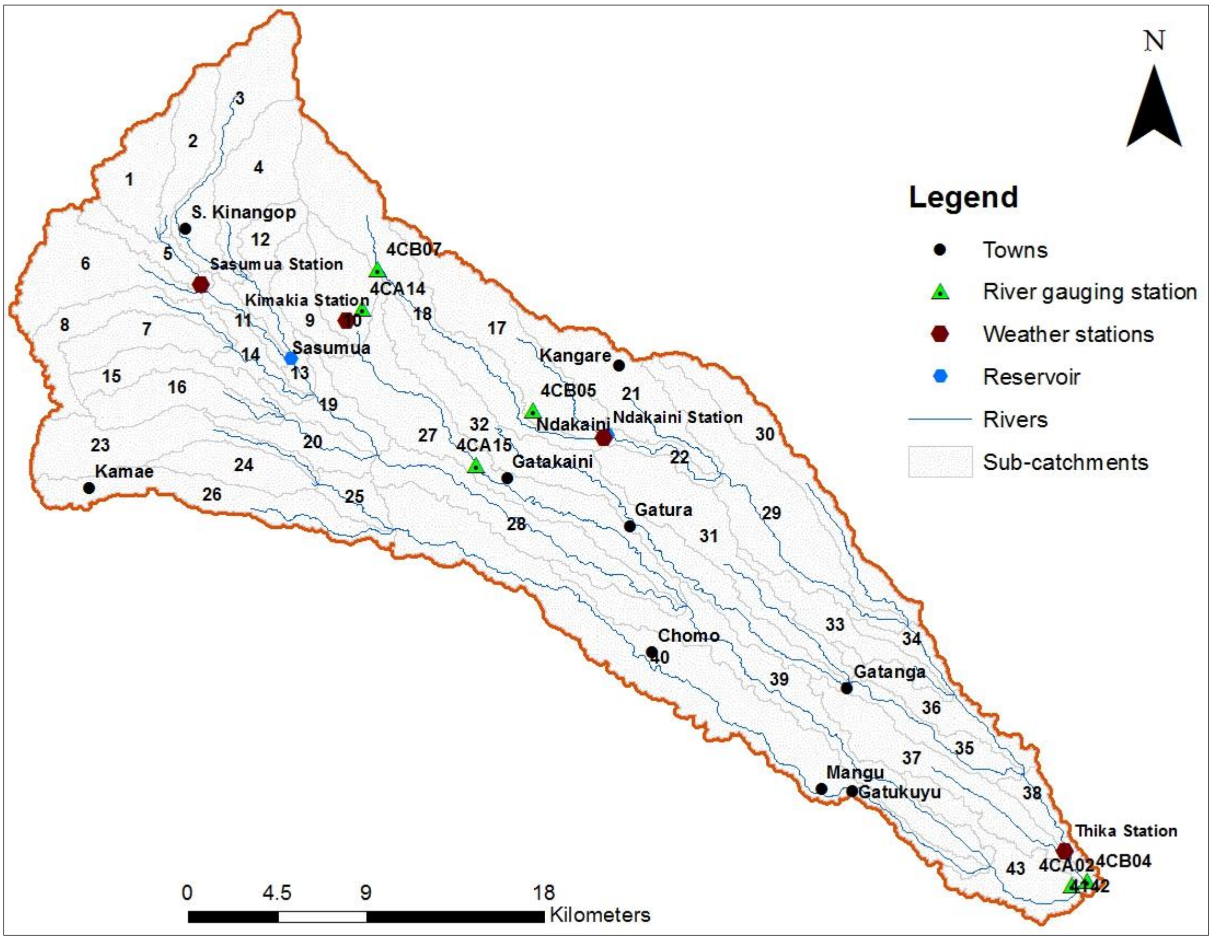

Figure 4.1 Thika-Chania sub-catchment, weather and river gauging stations

4.1.1 Sensitivity analysis

Global sensitivity analysis was achieved after 500 simulations using the SWATCUP SUFI-2 program and the results are presented in Table 4.1. Baseflow alpha factor (ALPHA_BF)), available soil water content (SOL_AWC) and the SCS runoff curve number(CN2) were found to be the most sensitive parameters in stream flow calibration.

According to Abbaspour (2015), the bigger the absolute value of the t-stat in global sensitivity analysis, the more sensitive the parameter is. The null hypothesis that a parameter is not sensitive is either accepted or rejected based on the p-values. Low p-values in Table 4.1 indicate that the respective parameter is sensitive to change in stream flow (Abbaspour, 2015).

Table 4.1 Parameters description and sensitivity analysis

Parameter	Description	t-stat	P-value
ALPHA_BF	Baseflow alpha factor (days)	23.48	0.00
SOL_AWC	Available water capacity of the soil layer	14.42	0.00
CN2	SCS runoff curve number factor	9.51	0.00
CH_N2	Manning's "n" value for the main channel	1.94	0.05
CH_COV2	Channel cover factor	1.22	0.22
SURLAG	Surface runoff lag time	0.93	0.35
USLE_K	USLE equation soil erodibility factor	0.69	0.49
GWQMN	Threshold depth of water in the shallow aquifer required for return flow to occur	0.26	0.79
GW_DELAY	Groundwater delay (days)	-0.22	0.83
GW_REVAP	Groundwater revap coefficient	-0.40	0.69

4.1.2 Calibration and validation of streamflow

Stream flow was calibrated after 5 iterations in SUFI2 program with each iteration having 500 simulations. Model calibration was conducted at gauging station 4CB05 between 1998-2005. In SUFI2, Abbaspour (2015) recommends at most five iterations in stream calibration. After every iteration, the parameters get smaller to fit in the region that produced better results in the preceding iteration. After the fifth iteration, the model enveloped 70% of the observed data within the 95 Percent Prediction Uncertainty (95 PPU). Figure 4.2 illustrates the monthly observed and simulated streamflow between 1998-2005. The model closely matched low flows but underestimated the high flows. Similar observation has also been observed in other studies using SWAT-CUP model (Meaurio *et al.*, 2015; Rostamian *et al.*, 2010). Tolson and Shoemaker (2004) reported that SWAT model doesn't not accurately predict the high flows events leading to underestimation or overestimation.

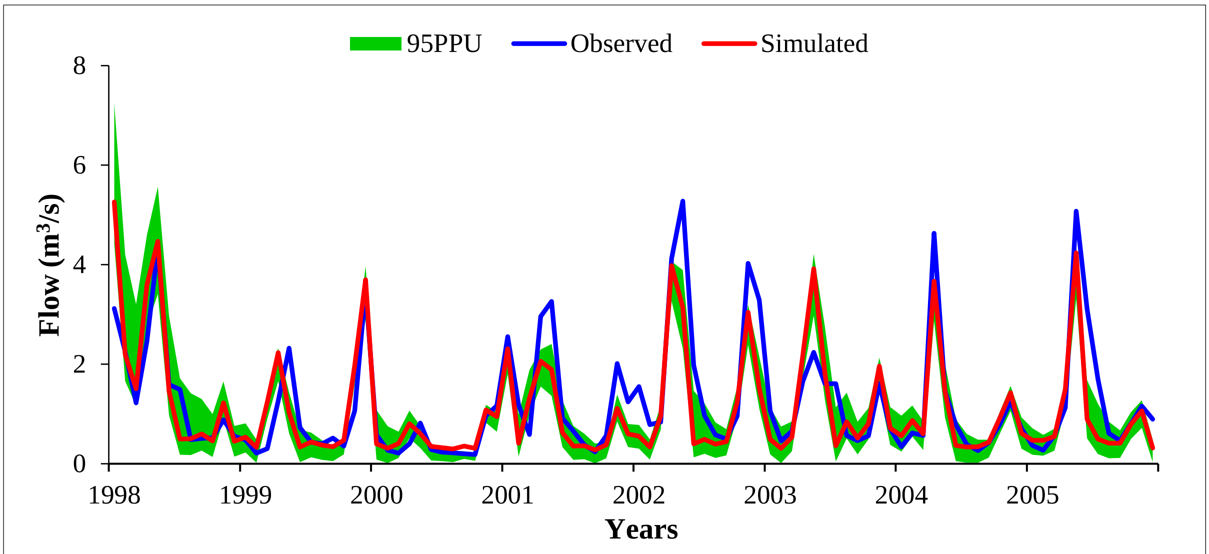

Figure 4.2 Monthly streamflow calibration results for gauging station 4CB05

The p and r factor were found to be 0.70 and 0.67, respectively. According to Abbaspour (2015), a p-factor greater than 0.70 and r-factor less than 1.5 are considered satisfactory for stream flow calibration. Therefore, the results of stream flow calibration in terms of p and r factor were acceptable. According to Abbaspour (2015), model calibration cannot be considered complete if the parameters uncertainty ranges are not reported. Therefore, the uncertainty ranges for the calibrated parameters are presented in Table 4.2.

Table 4.2 *Parameters qualifiers and uncertainty range*

Parameter Name	Qualifier	Fitted value	Min_value	Max_value
GW_DELAY	v	49.08	44.65	61.49
GWQMN	v	480.06	467.98	624.83
GW_REVAP	v	0.007	-0.14	0.42
USLE_K	r	0.38	0.27	0.44
SURLAG	v	15.49	6.96	16.18
CH_COV2	v	0.100	0.03	0.15
ESCO	r	-0.31	-0.60	-0.03
CH_N2	v	0.66	0.29	0.66
CN2	r	0.39	0.16	0.54
SOL_AWC	r	-0.18	-0.95	-0.09
ALPHA_BF	v	0.01	-0.10	0.11

*****r refers to the relative change of the specified parameter where the given value is added to one and then multiplied by the initial value of the parameter. v means that the initial parameter value is to be replaced by the given value.

The NS Efficiency (NSE), coefficient of determination (R^2) and the Percent BIAS (PBIAS) were found to be 0.66, 0.69 and 10.3, respectively. The model results are considered perfect when values of NSE and R^2 is one and that indicates a better match between the observed and the simulated results (Arnold *et al.*, 2012). However, according to Moriasi *et al.* (2007), the model can be judged as satisfactory if the NSE value is greater than 0.5 and PBIAS is within ± 25% for stream flow. If the NSE value is between the 0.65-0.75 and PBIAS value between ± 15-25, then the model is deemed good in simulating streamflow (Moriasi *et al.,* 2007). Considering the values of NSE and PBIAS in the current study, the model was good for stream flow calibration. An R^2 greater than 0.5 have been reported to represent a good match between the observed and the simulated flow (Moriasi *et al.,* 2007; Santhi *et al.,* 2001). Therefore, with an R^2 of 0.69 in this study, the model was considered acceptable.

The calibrated parameters and their uncertainty ranges were used for the validation of streamflow between 2006 and 2013 as shown in Figure 4.3. Similar to the calibration results, the validated results show high uncertainty with low flows than high flows as depicted by the 95 PPU in Figure 4.3. According to Abbaspour *et al.* (2007), the 95 PPU quantifies all uncertainties associated with the bracketed data. Therefore, the model was able to simulate the low flows correctly hence bracketing most of the data within the 95 PPU.

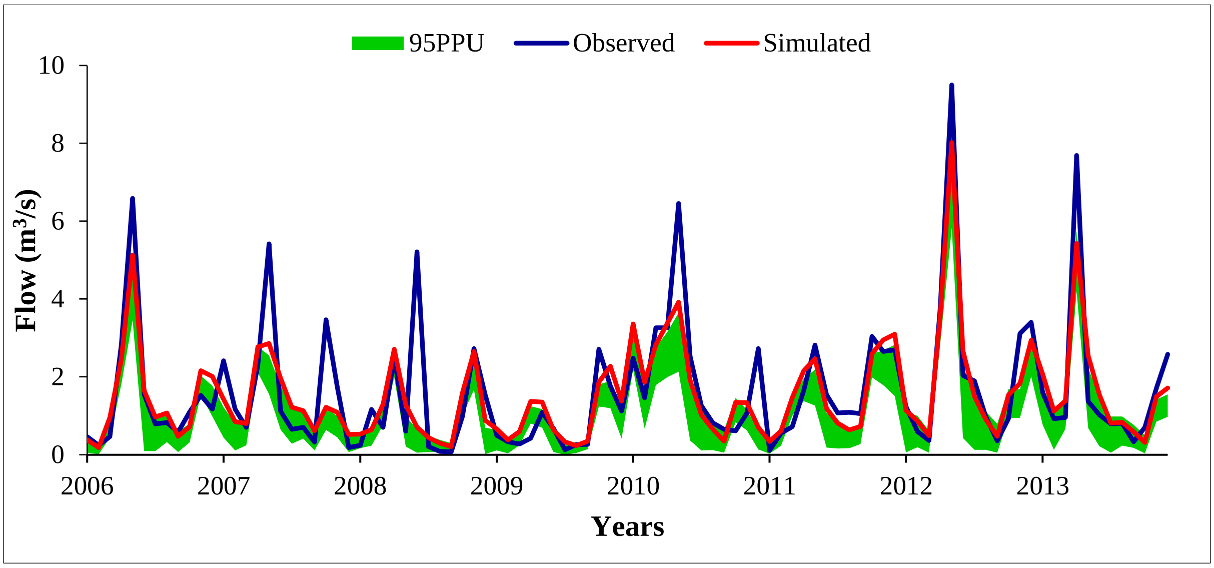

Figure 4.3 Monthly streamflow validation results for gauging station 4CB05

The NSE, R^2 and PBIAS were found to be 0.73, 0.75, and 7.2, respectively for the validation period. According to the thresholds given by Moriasi *et al.* (2007), the validation results were considered to be good for stream flow simulation. Other studies by Briak *et al.* (2016) observed an NSE value of 0.67 and PBIAS of -14.44 during model validation using SUFI2 and reported it as a good model performance. Results indicated that the p-factor and r-factor for the validation period as 0.61 and 0.45, respectively. Although the NSE, R^2 and PBIAS improved after validation, the p-factor reduced by 9 percent indicating that not as much data was bracketed within the 95 PPU as in the calibration process. This can be attributed to the data gaps and uncertainty associated with gap filling in the most recent data used in this study. However, according to Abbaspour (2015), the objective of calibration and validation is to have more data bracketed within the 95 PPU while keeping the r-factor to a minimum. In the current study, 61% of the observed data were incorporated within the 95 PPU hence the model results were considered acceptable. Rostamian *et al.* (2010) reported that low r-factor values indicate the goodness of the model calibration or validation. Therefore, the r-factor in this case portrayed better results during the validation process.

A scatter plot of the observed and simulated streamflow at River Gauging Station (RGS) 4CB04 is shown in Figure 4.4. An R^2 of 0.62 was obtained which implies that the model was able to simulate streamflow correctly both spatially and temporally. Begou *et al.* (2016) recommended the use of observed data in RGS not considered in calibration processes in order to make simulations more representative and realistic and also ensure reliable performance of the hydrological process within the catchment. According to Begou *et al.* (2016), the use of a different location for validation takes into account the heterogeneity of land use, climate, soils and terrain. The value of R^2 greater than 0.5 implies that the results of streamflow simulations are within acceptable ranges at different locations in the catchment (Santhi *et al.*, 2001).

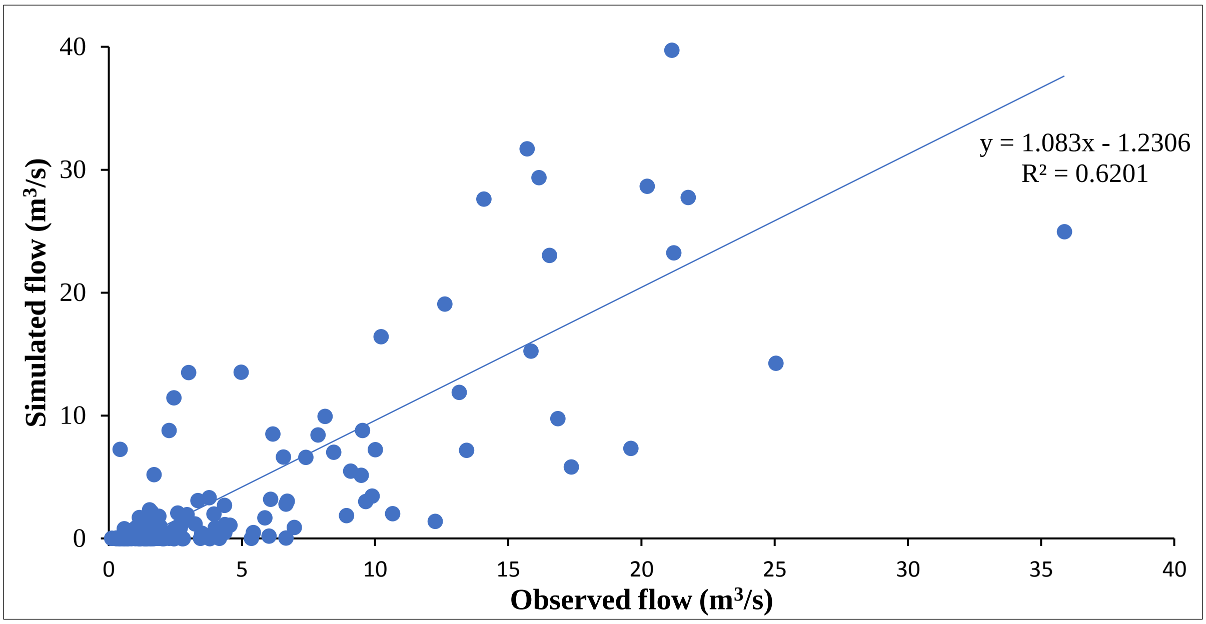

Figure 4.4 Observed and simulated stream flow at gauging station 4CB04

4.1.3 Calibration and validation of sediments

Sedimentation data from a reservoir bathymetric survey conducted by Hunink and Droogers (2011) was successfully used in the calibration and validation of sediments yield in the catchment. The survey indicated that the average sediment inflow into the immediate downstream Masinga reservoir is 5 Mton/year. A study conducted by Hunink and Droogers (2015) demonstrated that Thika-Chania catchment contributes 1.8 Mton/year of sediments inflow into the reservoir. These sediment yield was found to be equivalent to 21 tons/ha/year based on the total catchment area obtained in the delineation process. Therefore, Thika-Chania catchment contributes 36% of the total sediments inflow into Masinga reservoir. The results of the sensitivity analysis indicated that the channel re-entrained exponent (SPEXP), channel re-entrained linear parameter (SPCON), channel erodibility factor (CH_COV1), channel cover factor (CH_COV2), support practice factor (USLE-P) and the curve were sensitive to sediment. Abbaspour *et al.* (2007) reported similar results in a parameter sensitivity analysis for sediments using SWAT model in Thur watershed, Switzerland.

Table 4.3 Calibrated parameters for sediments in Thika-Chania catchment

Parameter Name	Qualifier*	Fitted value	Min_value	Max_value
CH_COV1	v	0.3	0.1	0.3
SPEXP	v	1.4	1.0	1.60
SPCON	v	0.005	0.001	0.005
CH_COV2	v	0.3	0.1	0.3
CN	v	50-88	36	98
USLE-P	v	0.95-1	0.80	1

*v implies that the parameter was directly replaced by the respective fitted value.

The mean annual sediment outflow from the catchment was compared with the simulated sediment yield. The results showed that the simulated annual sediment yield from the entire catchment is 21.507 t/ha. These results matched the observed sediment yield of 21.45 at the outlet of the catchment from the bathymetric survey. Therefore, the model accurately simulated the sediments yield at the catchment outlet.

The model was validated for sediments at Sasumua and Ndakaini dam which are located upstream of the Thika and Chania river confluence. It was found that the average annual sediments yield in Sasumua and Ndakaini dam was 0.60 and 0.67 t/ha. From the bathymetric survey, the two reservoirs were found to have approximately annual sedimentation rate of 0 Mt/ha. The results indicate although the model over predicted the sediment at the two reservoirs, the variation of the sediment yield across the catchment was captured. The results were therefore considered accurate given the uncertainty in input data which may include the effect of land use change (Vogl *et al.,* 2017) and reservoirs sediment trapping efficiency. The model results were further compared with those from a previous study conducted in Sasumua sub-catchment by Mwangi *et al.* (2015). The results of the current study indicated that the annual sediment yield in Sasumua sub-catchment was 11.0 t/ha. Therefore, this results agrees with the those of Mwangi *et al.* (2015) who indicated that the annual sediment yield the catchment was 10.3 t/ha. Similar methodology was used by Briak *et al.* (2016) who compared the sediments results of SWAT model simulation with those of previous studies conducted in Kalaya catchment in Northern Morocco. They reported that sediments yield results were similar to those of the past studies in the catchment. Thus, SWAT model can successfully be used to compare results of different studies in data scarce regions. Similarly the same approach has been used elsewhere in different studies by Hunink *et al.* (2013b) and Vogl *et al.* (2016).

The results of the calibration and validation of sediments showed that the model simulations were satisfactory in terms of simulating sediments across the catchment. This study sought to find out the reduction of sediments from the base scenario when soil and water conservation measures are implemented and not the actual sediments yield from the catchment. Therefore, the results of both sediments and streamflow calibration and validation were acceptable in this study.

4.1.4 Base scenario for simulation of soil and water conservation measures

The spatial distribution of the annual sediment yield within Thika-Chania catchment is presented in Figure 4.5. Analysis of the base scenario showed that the annual average sediment yield from the catchment is 21.507 t/ha. Sub-catchments where coffee, maize and general agriculture is practiced showed the highest sediments yield between 5-50 t/ha/year. Sediments yield were classified between high and very severe guided by Singh *et al.* (1992) classification. This study agreed with the results of Briak *et al.* (2016) and Cohen *et al.* (2006) who independently found that the highest sediments yield are observed in areas on agriculture and steep slopes. Areas on Tea were observed to have relatively low sediment yield which could be attributed from the extensive ground cover that controls the impact of rainfall and surface runoff on soil (Humberto and Rattan, 2008). Those areas with the largest percentage on forests had slight sediment yield ranging between 0-2 t/ha/year. The high sediment yield from sub-catchments number 30, 29 and 33 could be attributed to the type of land use (maize and coffee) and high slopes between 20-30%. The main land use in sub-catchment 30 is maize while sub-catchments 29 and 33 are on coffee. According to Hunink *et al.* (2013b), coffee and maize growing zones have been attributed to have the highest sediment yield. The results of the current study indicated that coffee and maize growing areas have an average annual sediment yield of ranging between 20 t/ha to more than 50 t/ha as shown in Figure 4.5.

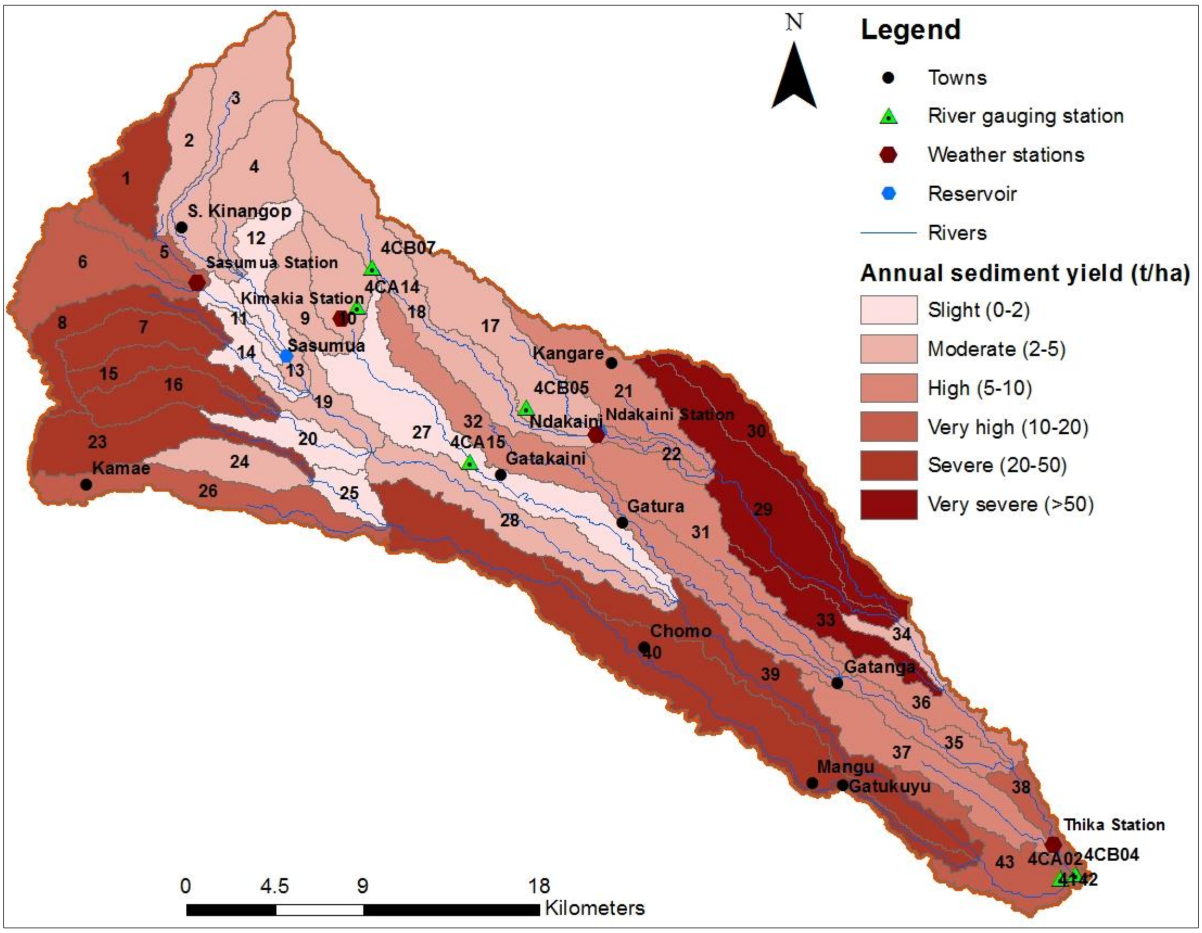

Figure 4.5 Spatial distribution of sediment yield across sub-catchments

The results of the base scenario analysis indicate that the total average annual water yield from the entire catchment was 922.36 mm. Analysis of the results showed that simulated surface flow was 202.28 mm while the baseflow was recorded as 638.20 mm. The annual average baseflow was higher than the surface flow which could be attributed to the well-drained soil in the catchment (NAAIAP, 2014). The average total water yield varied between seasons with the highest values observed during the rainy seasons in April/May and October/December as shown in Figure 4.6. The total water yield was also found to correlate with the rainfall and average sediment yield. Water yield in the dry season was observed to be low as the rainfall deceases in the months of June to September. This implies that soil and water conservation measures are needed to regulate sediment yield in rainy seasons and also enhance baseflow to recharge the surface flow in dry seasons. These results formed the base scenario for evaluating the impacts of soil conservation methods on water and sediment yield which represent the provisioning and regulating services of the ecosystem, respectively.

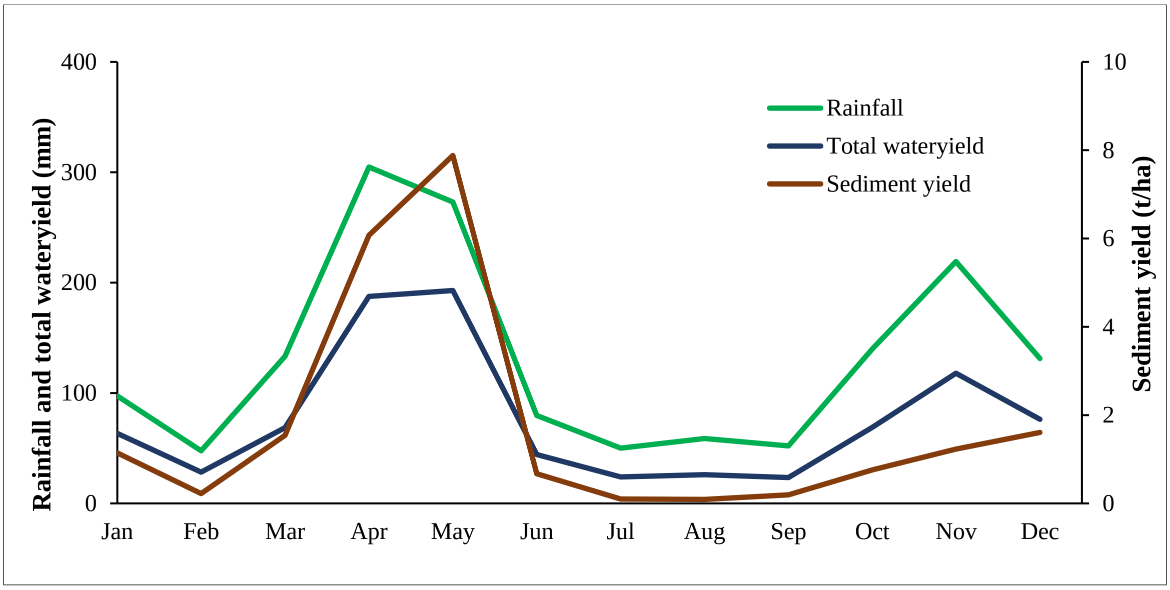

Figure 4.6 Monthly variation of sediment and water yield in Thika-Chania catchment

4.2 Simulation of Agronomic, Vegetative and Structural Conservation Methods

4.2.1 Impact of contour farming on sediment and water yield

Results indicated that contour farming reduced sediments yield by 35.81% from the base value of 21.507 t/ha/yr. Sub-catchments 1, 6, 8 and 7 indicated sediment yield reduction between 50-60% (Figure 4.7). These sub-catchments are mainly on general agriculture (e.g. cultivation of cabbages and potatoes) and bare/degraded lands. According to Mwangi *et al.* (2013), the sub-catchment are characterized by intensive farming of horticultural crops at small scale level thus practicing contour farming would be effective in improving the regulatory and provisioning ecosystem services. Similarly, sub-catchments 35, 36 and 37 that are on coffee, showed high percent reduction in sediment yield from the baseline simulation values. Other studies have reported similar positive impact of contour farming on sediment yield.

Gassman *et al.* (2006) in a study conducted in Northeast Iowa, indicated that when contour farming is practiced, sediment yields would be reduced by 34.0% based on a thirty-year annual average baseline results of SWAT model simulation. Shi *et al.* (2004) used the RUSLE model to study soil conservation planning in Wangjiaqiao catchment in China. They reported that contour farming with a seasonal no-till ridge would reduce soil loss rates by 70% from the status quo. Mwangi *et al.* (2015) assessed sediment yield reduction at the sub-catchment level where the annual base scenario was 8.98 t/ ha. They reported that practicing contour farming only reduced sediment yield from the base value to 7. 62 t/ha which represents 15.2% reduction.

Results of a study conducted by Tuppad *et al.* (2010) showed that contour farming as a conservation method reduced sediment yield from Bosque River catchment by 10% and 15.9% when implemented at the catchment and sub-catchment level, respectively. Tuppad *et al.* (2010) further reported that sediment reduction as a result of practicing contour farming could range between 28 to 67% at the HRU level. Therefore, contour farming is an effective method in controlling sediments yield and also improving crop productivity (Quinton and Catt, 2004) and hence the ecosystem services. Given that contour farming is not labor intensive and the cost of implementing is also low compared to structural practices (Phomcha *et al.*, 2012), the adoption rate by farmers could also be high. The results of the current study agree with the findings other studies that contour farming is an effective conservation method in reducing sediments yield and improving water quality.

Figure 4.7 represents the percent sediment reduction from base scenario value when contour farming is practiced in the catchment.

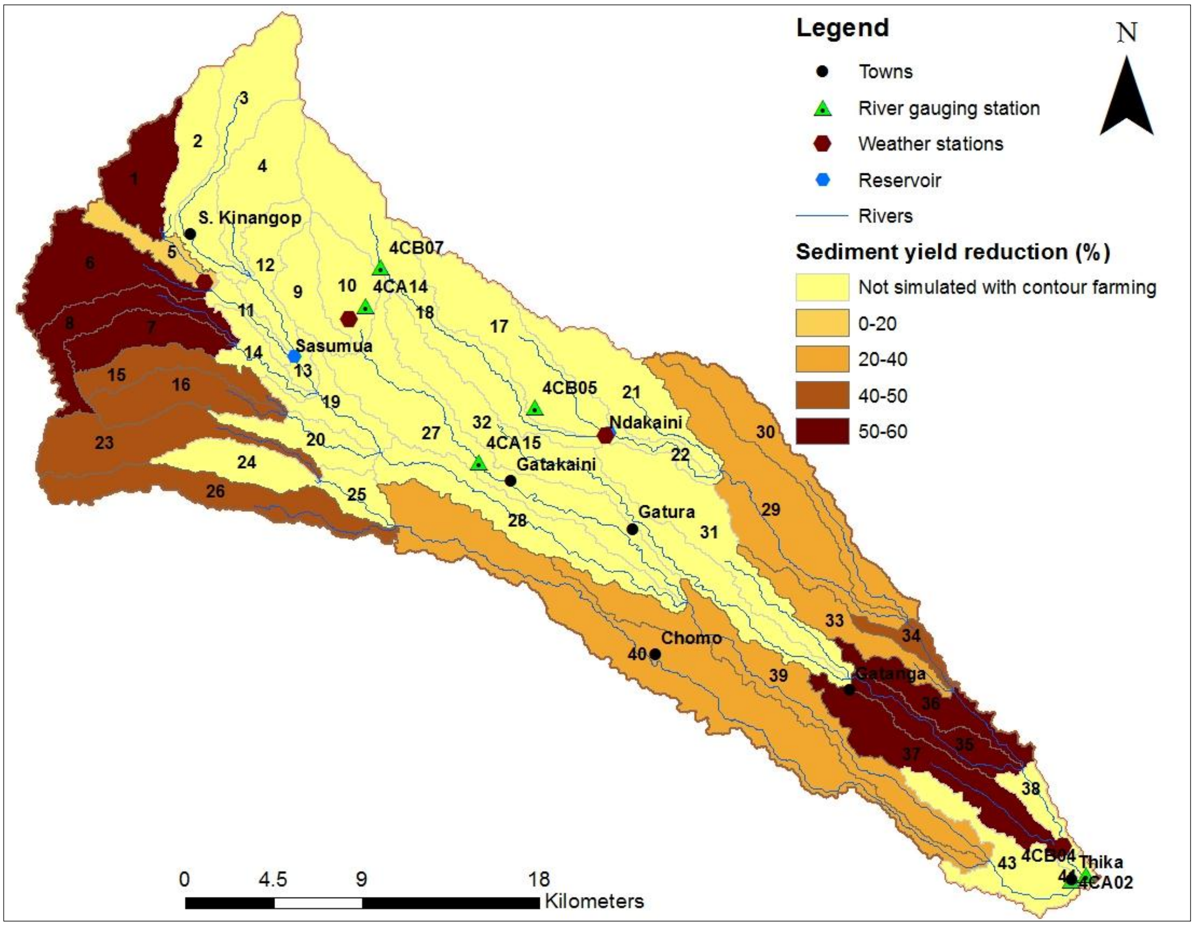

Figure 4.7 Percent sediment reduction after contour farming

Practicing contour farming in the catchment would reduce the total average annual water yield by 0.55% from the base value of 922.36 mm. However, the surface runoff generated from the catchment reduced by 16.40% from 202.28 mm while the baseflow increased by 3.38%. The flow in shallow groundwater aquifers increased by 8.01% from the annual mean of 76.75 mm. Increased baseflow indicated an increment of water in the shallow aquifer that would gradually be released to the streams and reservoirs during low flow seasons. This will enhance the provisioning ecosystem services in terms of water yield during dry seasons. On the other hand, reduced surface runoff implies that the amount of sediment and nutrients transported to the waterbodies will also reduce hence improving the water quality. Gassman *et al.* (2006) reported that implementation of contour farming would decrease the surface flow rate by 3.8% from the base SWAT simulation value. According to Morgan (2005), Tadesse and Morgan (1996), contour farming as a conservation method ensures water retention on the land by creating small depression and reducing the erosivity of the surface runoff. Therefore, contour farming also helps in minimizing the development of rills on the lands by facilitating water infiltration in the small depressions created during farming activities.

4.2.2 Impact filter strips on sediment and water yield

Simulation of filter strips in SWAT model improved the regulating services of the ecosystem by reducing sediment yield at the catchment outlet as shown in Figure 4.8. Simulation of a 3-meters filter strip width reduced sediment yield at the catchment outlet by 46%. Elsewhere, Robinson *et al.* (1996) observed that the rate of removal of sediment from a 3 meters width filter strip removed more than 70% of sediments surface runoff in northeast Iowa. Ramos *et al.* (2015) simulated the impacts of filter strips in controlling soil and nutrient losses in Anoia, Barcelona. They estimated that 3 m width filter strips would reduce soil losses by an average of 42%.

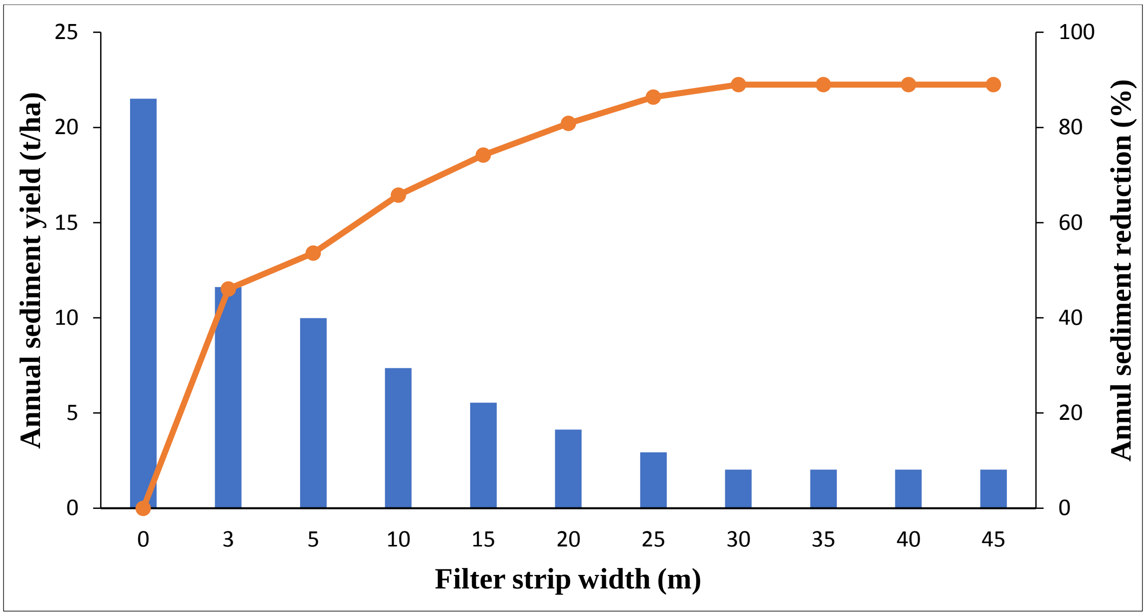

Figure 4.8 Annual sediment yield and percent sediment reduction at different filter width.

Filter strips of 5 meters width reduced the average annual sediment yield at the catchment outlet 53.57% from the average annual base scenario value of 21.507 t/ha (Figure 4.8). A 10 meters strip width on the other hand reduced sediment yield by 65.80% while a 30 meters strip width reduced the amount of sediment reaching the catchment outlet by 88%. The high efficiency of sediment removal in a 30 meters filter strip width could be attributed to the reduced flow rate in wider strips. Therefore, the effectiveness of reducing sediment yield from the catchment increases with increase in width of the filter strip. However, the amount of sediments trapped does not increase by bigger percentages beyond the 10 meters filter strip. Similar trends in sediment removal efficiency were observed in other studies by Yuan and Bingner (2009), Gharabaghi *et al. (*2004) and Parajuli *et al.* (2008). Gharabaghi *et al.* (2004) reported that most of the soil particles greater than 40 micrometers in diameter are removed within the first 5 meters of the filter strips. This can be attributed to the high sediment reduction within the first 3 meters of filter strip width in this study. According to Arabi *et al.* (2008), the filter width of a vegetative strip determines the trapping efficiency for sediments removal from surface runoff. Sediment characteristics, slope of the filter and the underlying soil characteristics also influence the trapping efficiency (Akan *et al.*, 2014).

Parajuli *et al.* (2008) explored the effectiveness of filter strips applied at the edge of fields in upper Wakarusha catchment in Kansas. They found that within the first 10 meters, sediments yields were reduced by 73% from the base scenario at the catchment level. From that study, Parajuli *et al.* (2008) reported that increasing the width of the filter strip to 10 and 20 meters reduced the annual average sediment yield at the catchment outlet by 82 and 89%, respectively.

A performance testing of vegetative filter strips conducted in Canada by Gharabaghi *et al.* (2004) revealed that the efficiency of sediment removal from overland flow varied between 50-98% with increase in filter strip widths. Helmmers *et al.* (2008) conducted a cost-benefit analyses of filter strips and found that filter strips reduced the annual sediment yield from 5.43 to 3.95 t/ha which represent 26% reduction. The reduction of sediments can be attributed to the reduced transport capacity of the surface flow through the filters hence deposition occurs. Moreover, reduced flow velocities and dense vegetation facilitates the trapping of the sediments (Akan *et al.,* 2014). Similar to the findings of Vaché *et al.* (2003), this study found that increasing the width of buffer strips beyond 30 m, the annual average sediment reduction would be insignificant compared to the narrower strips widths. Therefore, implementing a filter strips in the Thika-Chania catchment would still reduce a significant amount of sediments thus improving the regulating services of the catchment.

The provisioning ecosystem services in terms total average annual water yield of 922.36 mm from the base scenario were not influenced by the implementation of filter strips. The filter strips did not indicate any significant impact on either surface or baseflow. These results are consistent with the findings of Bracmort *et al.* (2006) who found that the surface runoff at the outlet of Dresbach and Smith Fry watershed were not influenced by filter strips. Bracmort *et al.* (2006) further stated that modelling of the conservation practice was only targeted at sediment yield reduction. According to Akan *et al.* (2014) the water infiltration in filter strips is influenced by permeability and the moisture condition of the soils. The main function of the filter strips is to trap sediment and fertilizers transported in surface runoff thus improving the water quality. Mwangi *et al.* (2013) evaluated the impacts of vegetative filter strips on ecosystem services and reported insignificant change in total water yield at the catchment outlet. The representation of filter strips in SWAT does not include adjusting the curve number which control the surface runoff volume (Arabi *et al.,* 2008) which can be attributed to the insignificant change in runoff volume. According Arabi *et al.* (2008), filters strips are installed to reduce sediments, fertilizers and bacteria in surface runoff as it passes through vegetation cover. Gharabaghi *et al.* (2004) tested the performace of vegetative filter strips and found that smaller sediment aggregates could only be removed through the infiltration of water. However the results of the present study does not indicate infiltration runoff water as would be potrayed by increased baseflow or reduced surface runoff depth. Further studies are needed to determine the impacts of filters strips on the infiltration capacity of the soil and hence the impact on ground water yield.

According to the national land use guidelines for Kenya, a minimum of two meters should be left on both side of the river as riparian area (NEMA, 2011). The guidelines further stipulate that a minimum of 30 m should be left uncultivated from the highest water mark during peak flows. As such, if these guidelines are followed, then the amount of sediments reaching waterbodies would significantly be reduced. Institutional strengthening and capacity building are some of the initiatives that should be undertaken to enhance proper functioning of the ecosystems.

4.2.3 Impact of terraces on sediment and water yield

Terraces were simulated in all HRUs except in forest and tea zones and areas where the average HRU slopes were more than 2.3%. According to Arabi *et al. (*2008), reducing slope length in HRU where the slope is less than 2.3% would result in higher erosion estimate. Slopes analysis in forested and tea zones areas indicated steep slopes ranging from 30% to more than 50% and therefore would not be feasible to implement terraces (USDA NRCS, 2015).

Implementing terraces in Thika-Chania catchment reduced sediment yield by 80.7 % from the annual average base scenario value of 21.507 t/ha at the catchment outlet. Figure 4.9 shows the percent reduction of sediment at each of sub-catchments where terraces were implemented.

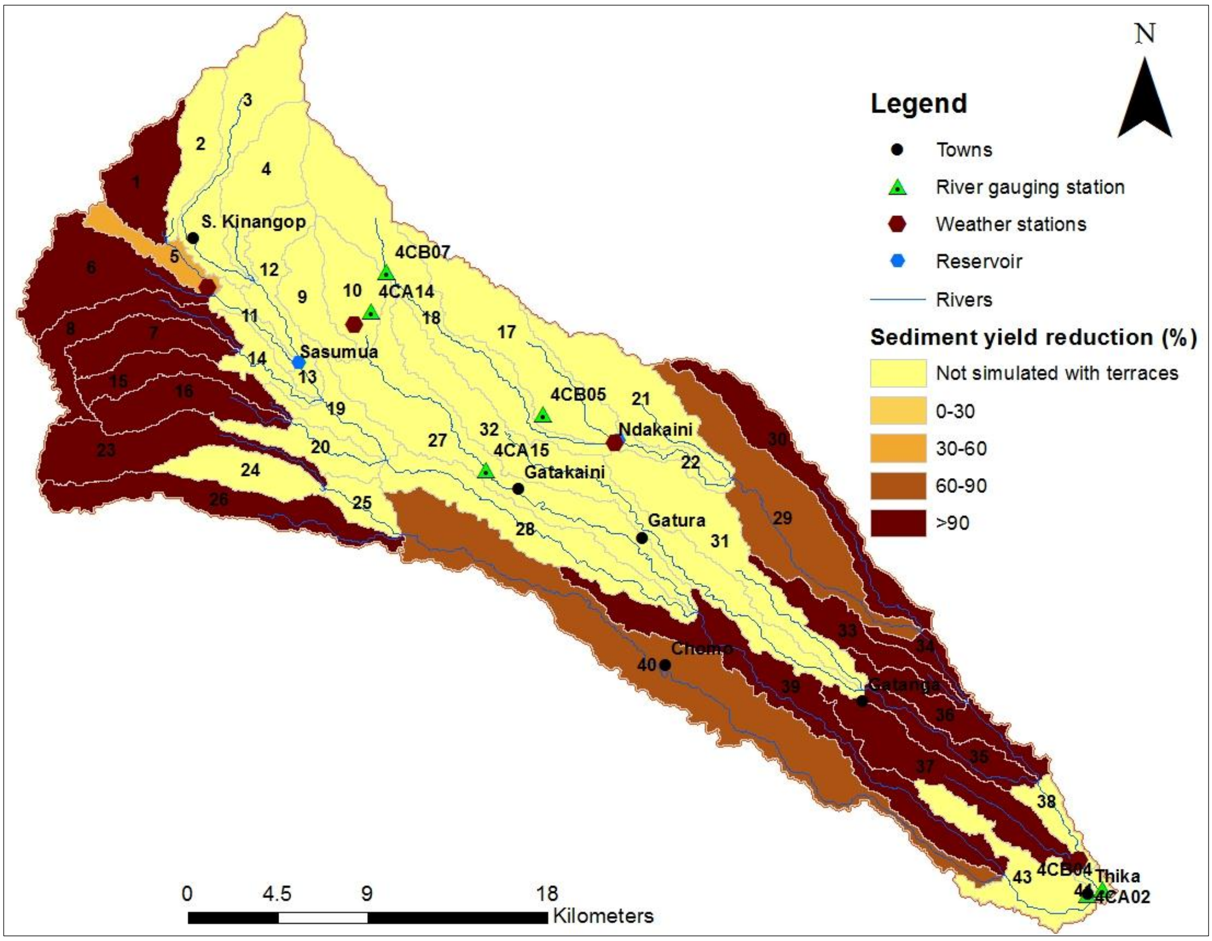

Figure 4.9 Percent sediment reduction after terracing

Sediments yield reduction above 50% was observed in areas on coffee and maize and gen agriculture (which includes areas on vegetables and other horticultural crops). Areas that have steep slopes ranging from 20-30% exhibited high sediment reduction when terraces are practiced. However, sub-catchments 43, 38 and 5 had sediment yield reduction of less than 50% from the respective base scenario value. This was attributed to the presence of rangelands in the three sub-catchments where terracing was not simulated. Other studies for instance Gassman *et al.* (2006) evaluated alternative methods for sediments and nutrients in northeast Iowa and found that terraces provided the greatest sediment loss reduction of more than 60% from the prevailing conditions. Another study by Tuppad *et al.* (2010) reported that sediments yield can be reduced by a magnitude of between 57 and 95% at the HRU level. At the outlet of the catchment, Tuppad *et al.* (2010) found that the average annual reduction in sediment yield was 17.2%. Arnáez *et al.* (2015) reviewed the effects of terraces on hydrological processes and found that terraced areas in semi-arid and semi-humid areas can have annual sediment yield below 3 t/ha. In this study coffee and maize areas were observed to have sediment yield ranging between 20-50 t/ha. Therefore, terracing those areas would reduce the amount of sediment reaching the waterbodies and generally improving the ecosystem services.

Results showed that terraces are more effective in sediment yield reduction than filter strips and contour farming. However, an economic model simulation by Gassman *et al.* (2006) indicated that terraces are the most expensive conservation practice to implement. If terraces are implemented, they would have the greatest benefit in terms of sediment reduction from uplands per unit area. The greatest beneficiary of terraces is the coffee growing zones where erosion rates were modelled to be the highest. Terraces reduce the slope lengths thereby decreasing the runoff velocity thus reducing the amount of sediments in surface runoff.

Surface runoff was reduced by 30.25% from the base value of 202.28 mm while baseflow was increased by 8.38%. Shallow aquifers recharge increased by 5.08% from 76.75 mm. Although the flow components were observed to change, the total water yield from the catchment reduced by only 0.40%. This shows that terraces would be very useful in reducing flashfloods associated with peak surface runoff. By increasing the amount of runoff retained within the terraced fields, the amount of water available to recharge shallow groundwater aquifers is increased. However, when surface runoff is used to feed reservoirs, then other methods that does not significantly reduce the surface water should be implemented. Nevertheless, implementing terraces only on some sections of the catchment, the effect on runoff volume would be negligible. Gassman *et al.* (2006) practiced terracing on a 2% of the catchment area and found that their effect on streamflow was insignificant. Figure 4.9 shows the sub-catchments within which terraces could be implemented to achieve the highest percent reduction in sediment yield. These results also show that terraces can be used both as control for soil erosion and peak runoff rates.

Terraces cause ponding of water on the flat surfaces during rainfall events hence increasing infiltration. According to Arnáez *et al.* (2015), terraces enhance infiltration of water into the soil hence reducing the surface runoff and increase baseflow. The results of this study are also consistent with the findings of Schmidt and Zemadim (2015) who observed that terraces implemented on slopes greater than 5% could reduce sediment and water yield by 90% and 43%, respectively. Gassman *et al.* (2006) estimated that terraced areas reduced flow rates by approximately 4% at the outlet of the catchment from the base value. Therefore, reducing flow velocity can reduce the erosivity of the surface runoff hence less sediments are generated. Successful implementation of terraces in Ethiopia has been reported by Lemann *et al.* (2016).

According to Lemann *et al.* (2016), terraces implemented on 50% crop lands reduced sediments yield by approximately 30% in upper Blue Nile catchment, Ethiopia. They concluded that there is still the potential to reduce sediment further by terracing more areas in the catchment.

4.2.4 Impact of grassed waterways on sediment and water yield

Results of grassed waterways simulation in seasonal tributaries and channels conveying water from the cultivated lands indicated that sediment yield can be reduced by 53.90% from the base scenario value of 21.507 t/ha/yr. at the outlet of the catchment. Figure 4.10 represents the percent reduction of sediment yield from the status quo at sub-catchment level. This information could be used by water managers and catchment management officers to prioritize areas for implementing grassed waterways. The current study indicated that sediment removal from surface runoff into Ndakaini reservoir (outlet of sub-catchment 17) would be reduced by 56.70% while the yields from Sasumua catchment (outlet of sub-catchment 11) would be reduced by 38.06% if grassed waterways are implemented. This shows that the practice is an effective way of reducing the amount of sediments in surface runoff which is achieved by creating channel roughness hence facilitating deposition. Grassed waterways can be used to convey surface runoff that cause erosion and/or flooding from low-lying areas that experience ponding. They can also be used to effectively discharge runoff from urban centers. The results of the present study imply that if implemented and maintained properly, the effectiveness of grassed waterways would be improved hence the annual average sediment yield from the catchment would decrease. Ultimately, regulating and provisioning services of the ecosystem in the catchment will be restored and improved. More provisioning services like cattle fodder are obtained from vegetation along the waterways. Elsewhere, a study conducted by Gassman *et al.* (2006) found that grassed waterways reduced sediment yield in upper Maquoketa catchment by 45.9% from the prevailing condition. The results of the current study are consistent with those of Bracmort *et al.* (2006) who observed that grassed waterways had an impact of sediment and phosphorous yield at the outlets of Dresbach and Smith Fry catchments. Bracmort *et al.* (2006) showed that grassed waterways reduced sediment and phosphorous yield by This implies that implementation of grassed waterways will also impact on nutrients yield in the catchment by retaining them in the field. This will enhance soil fertility and hence increased productivity. Mwangi *et al.* (2015) estimated the impact of grassed waterways and found that they reduced sediment yield by 54% from the base SWAT simulation value in Sasumua catchment.

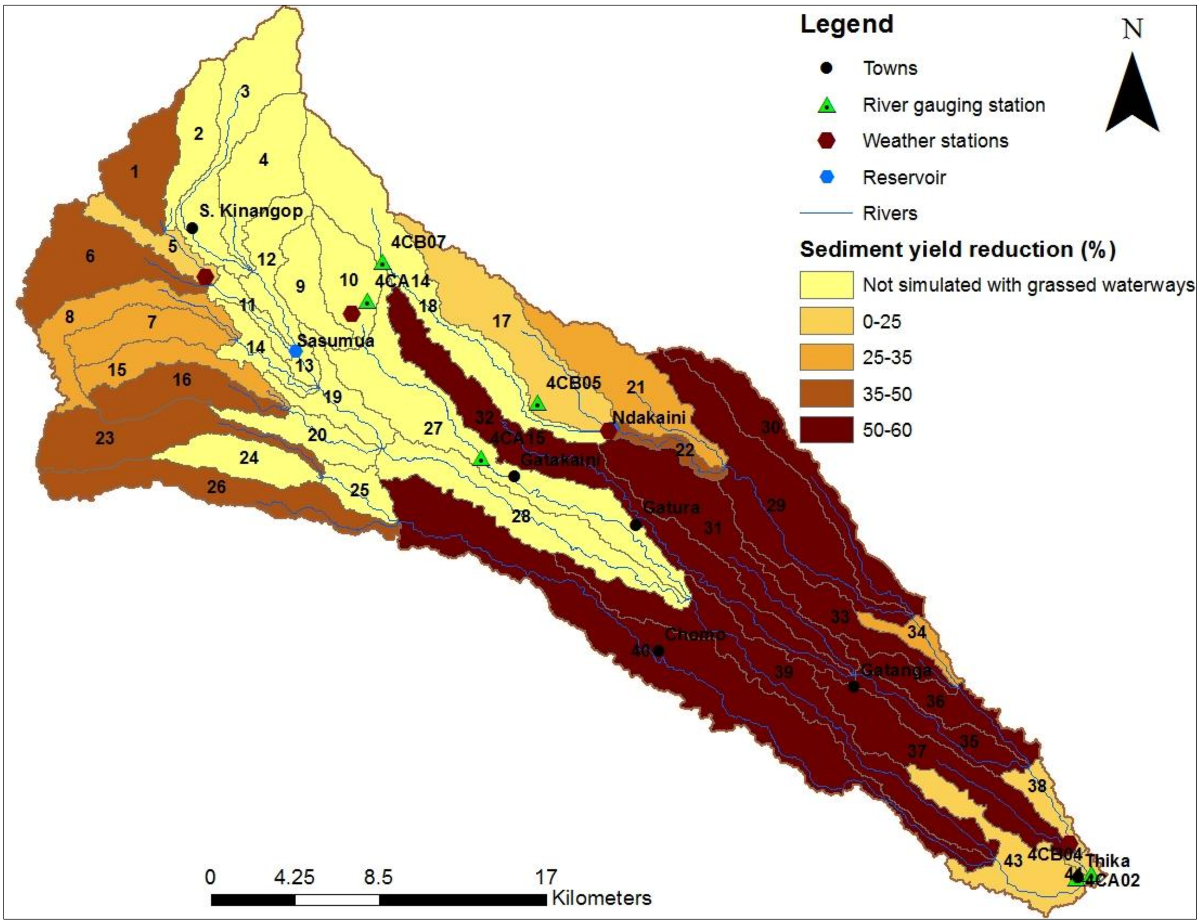

Figure 4.10 Percent sediment reduction after implementing grassed waterways

The annual average surface runoff showed no change from the base scenario hence the total water yield did not change. The results of water yield and sediment yield are summarized in Table 4.4. The results indicate that grassed waterways had no impact on the water yield and therefore can be implemented upstream of Ndakaini and Sasumua reservoirs that mainly depend on surface runoff. A study conducted by Gassman *et al.* (2006) reported that grassed waterways can reduce flow rates of the surface runoff thus enhancing sediment deposition. They estimated that the total flow rate reduced by 3% from the initial scenario after implementing grassed waterways. The results of the this study also agrees with those of Mwangi *et al.* (2015) who reported that grassed waterways had no impact on surface runoff and the average water yield from the catchment no significant change in total water yield was observed. They observed that grassed waterways showed greater impact on erosion reduction and sediment deposition along the waterways. Similar results were obtained by Bracmort *et al.* (2006) who estimated that grassed waterways had no impact on surface runoff volume and streamflow at outlet of Dresbach and Smith Fry catchment. Fiener and Auerswald (2005) used experimental field data to determine the impact of grassed waterways on runoff. Contrary to the results of the SWAT modelling, Fiener and Auerswald (2005) observed that depending on cross-sectional area of the thalweg, grassed waterways reduced surface runoff between 50 to 90%. The contradicting results could be attributed to using a different approach of modelling where lateral flow was minimal or excluded in this study. Although the results of this study agrees with the findings of Fiener and Auerswald (2005) that grassed ways reduced sediment yields, more research need to be conducted to compare experimental field results and those of SWAT modelling on the impact of grassed waterways on water yield in the same agro-climatic zone.

Table 4.4 *Sediments and water yield in grassed waterways*

Variable		Base-value	Simulated	Change (%)
Sediments (t/ha/yr.)	Thika-Chania outlet	21.51	9.90	-53.90
	Ndakaini dam	0.67	0.29	-56.70
	Sasumua sub-catchment	11.00	6.81	38.06
Water yield (mm)	Surface runoff	202.28	202.27	-0.005
	Baseflow	638.20	638.20	0
	Shallow aquifer recharge	76.75	76.75	0
	Total water yield	922.36	922.36	0

4.3 Combination of Different Conservation Measures

4.3.1 Contour farming and filter strips

Based on the average land sizes in the catchment (Leisher, 2013) and also the integrated land use guidelines (NEMA, 2011) it was assumed that a 3 m filter strip would be implemented on wide range in the catchment. A study by Kyalo *et al.* (2014) found that the most popular combination of soil and water conservation practice by smallholder farmers was contour farming and grass strips.

Practicing contour farming together with a 3-meters filter strip would reduce sediment yield at the catchment outlet by 62.8% from the base average annual value of 21.507 t/ha. This shows that the combination further increased sediment reduction by 27% compared to implementing contour farming individually. A 3-meters filter strip reduced 46.04% of sediments produced in the catchment but when combined with contour farming, 16.76% more reduction was achieved.

These results indicate that combination of the two conservation practices can significantly increase sediment yield reduction. However, after integrating the conservation method, the total water yield for either practice remained unchanged thus the combination has no impact on water yield of individual practices. Similar observation was reported by Bracmort *et al.* (2006) that combination of filter strips with other conservation measures had no impact of surface runoff. Although the combination did improve the provisioning services of the individual practices in terms of water yield, the regulation services were greatly improved.

The integrated national land use guidelines further stipulate that a minimum of 10 m filter strip could be left between areas with intensive activities like quarries and rivers. Therefore, a 5 m and 10 m filter strip width were combined with contour farming to assess their impact on sediment reduction. Results indicated that 5 m filter strip and contour farming combination reduced sediment yield at the catchment outlet by 67.26% while 10 m filter strip combination reduced sediments by 74.46%. This shows that increasing filter width beyond 10 m and integrating them with contour farming, sediment yield can be reduced by more than 70% from the base scenario value. Similar observations were made by Mwangi *et al.* (2013) who reported that contour farming implemented together with 5 meters filter strips reduced sediment loading by 73% from the baseline value. Another study by Mwangi *et al.* (2015) evaluated the impact of a 10 m filter strip combined with contour farming and reported sediment reduction by 41% on catchment of approximately 110 km^2. When contour farming was practiced alone, Mwangi *et al.* (2015) recorded sediment reduction of 24%. Threfore from these analysis, contour farming and filter strips can effetctively be used to restore and improve the ecosystem services.

4.3.2 Filter strips and terraces

A survey conducted by Leisher (2013) reported that terraces and grass strips are common soil and water conservation measures in Thika-Chania catchment. Thus, the impact of combining filter strips and terraces on water and sediment yield was evaluated.

Filter strips of 3 m width were combined with terraces and implemented in areas on coffee plantation, general agriculture, maize and bare/degraded lands. Results show that combining terraces and 3 m filter strip reduce sediment yield by 84.92% compared to 80.7% when terraces are practiced alone. This indicates an increment of 4.22% sediment reduction which implies that terraces could effectively be used to reduce sediments yields without integration with other practices. However, filter strips (e.g. vetiver grass) should be planted along the edges of the terrace to increase its stability (Humberto and Rattan, 2008). Elsewhere, Ramos *et al.* (2015) simulated the impacts of soil conservation measures on soil and nutrients losses in Piera, Barcelona. They noted that combination of terraces and filter strips reduced soil losses by 57% compared to an average of 50% when terraces are practiced alone. Ramos *et al.* (2015) concluded that introduction of the two measures significantly reduced nutrient losses as well.

Implementing 5 m width filter strips and terraces reduced sediment yield by 85.61% while a combination with a 10 m width filter strips reduced sediments by 86.73%. The results indicate that increasing the filter width in areas fully implemented with terraces does not greatly change the amount of sediment reduced. The study by Mwangi *et al.* (2015) demonstrated that 10 m filter strips combined with terraces reduced sediment yield by 43% compared to 35% when only filter strips are used. This represents an increment of 7% sediment reduction. The current study indicated that when 5 m filter strip combined with terraces is increased to 10 m, sediment reduction is increased by 1.12%.

The surface runoff from these combinations was reduced by 30.25% which did not reflect the impact of filter strips on water yield. Similar reduction in surface runoff was recorded when only terraces are practiced. Therefore, combination of the two methods does not influence the water yield of the individual practice but impacts significantly on sediment yield. Results of the current and previous studies show that terraces and filter strips can reduce sediment and nutrient yield generated from a unit area. This eventually improves water quality hence the ecosystem services are restored. A cost-benefit analysis of these practices would help determine which combination would be useful to optimally and sustainably reduce sediment yield at the catchment or sub-catchment level.

4.3.3 Grassed waterway and terraces

Combining grassed waterways and terraces was found to have an influence on water and sediment yield from the catchment. Results showed that sediment yield reduced by 88.72% from the annual average of 21.507 t/ha. This depicts an increment of 8% from 80.7% sediment reduction when terraces are implemented alone. Based on these results, grassed waterways did not greatly impact on sediment yield when they are combined together with terraces. However, Fiener and Auerswald (2006b) recommended the use of grassed waterways to discharge surface runoff from terraces and other water concentration points that may cause erosion and flooding.

The average surface runoff from the catchment decreased by 30.25%. This value however does not reflect the influence of grassed waterways on water yield. Implementation of terraces have shown that they can control both water and sediment yield from a given area hence impacting on both regulating and provisioning services. Grassed waterways on the other hand are effective in slowing down runoff thus facilitating deposition of sediments by enhancing the channel roughness. The baseflow and shallow aquifer recharge remained unchanged from the terrace simulation percentages of 8.38% and 5.08%, respectively. Similar results were observed by Mwangi *et al.* (2013) where the impact of grassed waterways on water yield was negligible but significant sediment reduction was reported.

4.3.4 Grassed waterway and contour farming

Combination of grassed waterways with contour farming reduced sediment yield by 70% from the annual average of 21.51 t/ha at the catchment outlet. Sediment reduction increased by 34.19% from 35.81% when contour farming is implemented alone. On the other hand, sediment reduction increases by 16.03% from 53.97% if grassed waterways are implemented on their own. These results are consistent with a study conducted by Mwangi *et al.* (2015) to assess the impacts of grassed waterways and contour farming combination on sediment yield reduction. They found that when implemented together, the two conservation methods reduced sediment yield by 66% from the base value. According to Mwangi *et al.* (2015), the combination improved sediment reduction by 42% compared to when contour farming is practice alone. These results show that grassed waterways and contour farming is an effective combination to reduce sediment yield from the catchment. Nutrients carried in surface runoff are also trapped thus water quality is improved while ensuring that the fertilizers remain in the agricultural fields. The vegetation planted along the waterway could also be used as cattle fodder thus the ecosystem services are increased.

The average annual surface runoff was not influenced by combining both practices. Surface runoff reduction of 16.40% similar to when contour farming is practiced was recorded. The results indicated that grassed waterways had no impact on total water yield from the catchment even after combining with other practices. The reduction in surface runoff when contour farming is practiced can be attributed to the small depressions and surface roughness created during cultivation (Stevens *et al.*, 2009). Bracmort *et al.* (2006) validated the results of SWAT modelling in two catchments and reported that grassed waters did not have an impact on water yields in the two catchments. According to Evrard *et al.* (2007), grassed waterways reduced the velocity of the surface runoff and in some case may facilitate infiltration. Enhanced infiltration would recharge the baseflow and reduce surface runoff hence influencing the total water yield from the catchment. Nonetheless, grassed waterways and contour farming will collectively restore and improve the regulating and provisioning services of the ecosystem. Vegetation along the grassed waterways could also be used as cattle fodder during the dry seasons.

4.3.5 Grassed waterways and filter strips

The current study showed that combining grassed waterways and 3 m filter strips reduced annual average sediment yield by 74.55%. The effect of the combined practices was found to be better than the sediment reduction of individual practice. Sediment reduction improved by 28.5% from 46.04% when filter strips are combined with grassed waterways. Similarly, the effectiveness of the grassed waterways in sediment reduction is further improved by 20.58%. Therefore, combining grassed waterways and filters strips is an effective way of improving ecosystem services in Thika-Chania catchment.

Further evaluation of grassed waterways with a 5 m and 10 m filter strip indicated a reduction of annual average sediment yield by 77.72% and 82.86%, respectively. Integrating grassed waterways with 10 m filter strips increased yield reduction by 7.31% compared to 3 m filter strip integration. A 3-meter filter strip integration with grassed waterways can be recommended for implementation in small holder agricultural lands and waterways installed along the roads that discharge water to the agricultural fields. Mwangi *et al.* (2015) evaluated the impact of combining grassed waterways and filter strips of different width. They reported that 10 m filter strips combined with grassed waterways reduced sediment yield by 73% from the baseline value. Mwangi *et al.* (2015) further tested the impact of combining grassed waterways with 30 m filter strips. They observed 80% sediment yield reduction which translates to 7% increment from the combination using 10 m filter strips. This implies that when resources are limiting e.g. land or financial resources either of the methods can be used to reduce sediment yield from a given area.

This combination helps in improving water quality due to reduced contaminants in the surface runoff. According to Mutegi *et al.* (2008), vegetation used in filter strips or grassed waterways e.g. napier grass have been found to reduce soil erosion and can be effective one year after establishment. Implementation of filter strips and grassed waterway improves the regulating services of the ecosystem by filtering out sediments and nutrients from the surface runoff. The provisioning services are also improved when the grasses grow and can be used as animal fodder (Mwangi *et al.,* 2013). In some areas, the grass is used for mulching thus preventing water loss through evaporation, maximizing crop yields and growth (Kwambe *et al.,* 2015). In the latter scenario, the supporting services are provided.

4.3.6 Grassed waterways, terraces, filter strips

Leisher (2013) reported that terraces, filters strips, cover crops and contour farming were the most common soil and water conservation practices in Thika-Chania catchment. However, in their report, Leisher (2013) grassed waterways were not reported either because they are not practiced on farmlands or were not assessed in that study. This combination sought to evaluate the impact of introducing grassed waterways, terraces and filter strips on water and sediment yield from the catchment. The combination also sought to have filter strips implemented in steep areas where terraces construction would not be feasible. For example, installing terraces and grass strips on coffee areas and maize areas (which recorded highest erosion) and grassed waterways along the valley to collect runoff from the upslope. Areas on forests were not simulated with soil and water conservation practices.

Implementation of grassed waterways, terraces and 3 m filters strips reduced annual average sediment at the catchment level by 90.42%. The combination improves the sediment yield reduction by 9.72 compared to when terraces are implemented individually. These practices would be highly effective in coffee and maize growing areas where soil erosion is high and steep. The average annual surface runoff reduced by 30.25% while the baseflow reduced by 8.38%. However, the effect on water yield is only a function of terraces as filter strips and grassed waterways had no influence on the total water yield. These results agree with conclusion of Bracmort *et al.* (2006) who showed that only terraces had an impact on surface runoff in a grassed waterways, filter strips and terraces combination. However, Bracmort *et al.* (2006) reported annual average sediment reduction by 32% from 0.68 t/ha to 0.46 t/ha in Dresbach catchment.

Figure 4.11 shows the percent reduction of sediment yield after implementation of the combined conservation practices. At the sub-catchment level, sediment reduction by more than 60% was achieved. Majority of the sub-catchments on coffee and maize recorded more than 90% reduction in sediment yield from the baseline value. Mwangi *et al.* (2015) evaluated the impacts of combining terraces, grassed waterway and 10 m filter strip on sediment yield reduction from the baseline condition. They reported 75% sediment reduction when the practices are implemented together. These practices are therefore effective both at the catchment and sub-catchment level. Their cost-effectiveness need to be established so that feasible conservation measures with the optimal effects on ecosystem services are implemented.

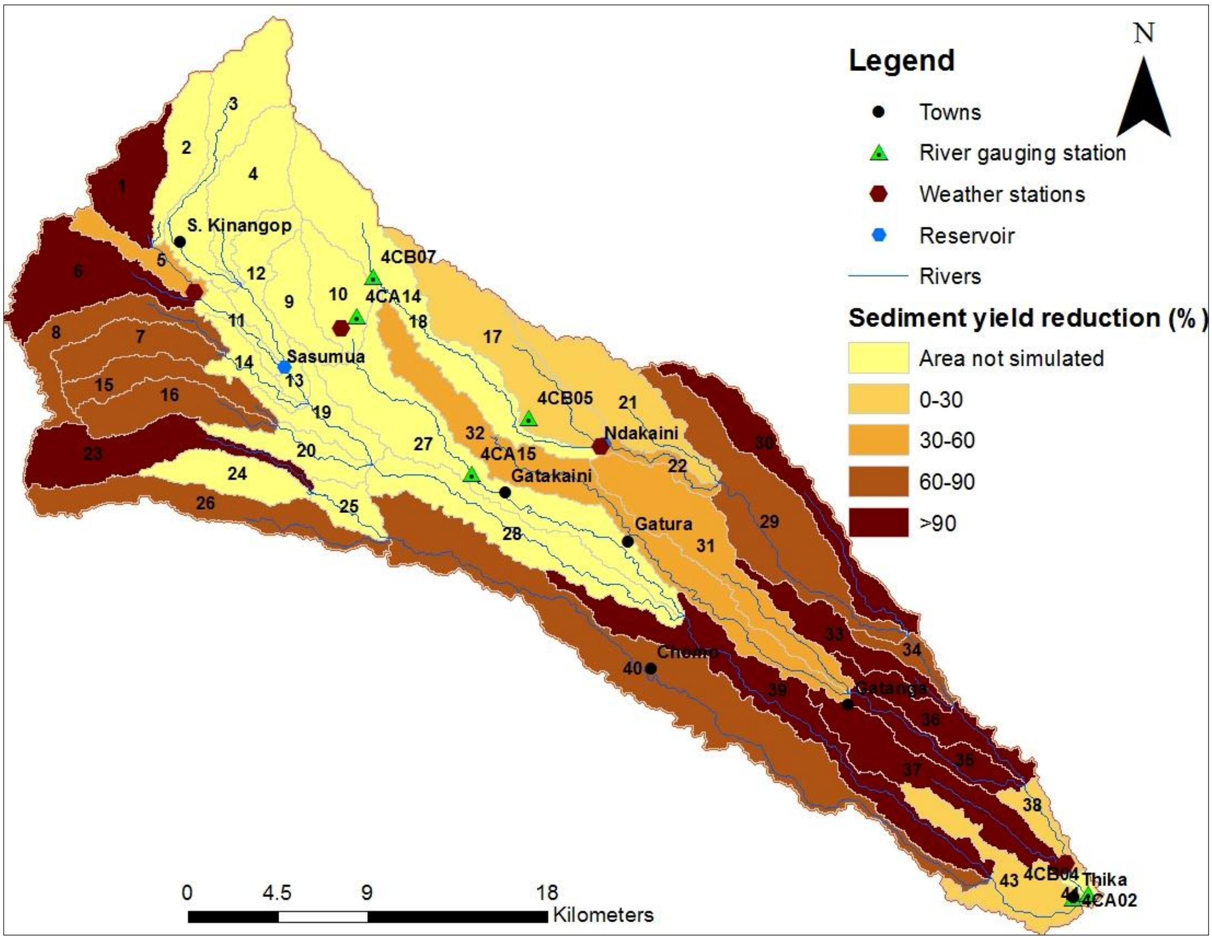

Figure 4.11 Percent sediment reduction after integration of GWW, filter strip and terraces

CHAPTER 5
CONCLUSION AND RECOMMENDATIONS

5.1 Conclusion

SWAT model was effectively applied to evaluate the impact of soil and water conservation practices on ecosystem services that included water and sediment yield. Parameter sensitivity analysis indicated that ALPHA_BF, SOL_AWC, SCS curve number, and CH_N2 were the most sensitive parameters to streamflow. Five parameters that were found to be most sensitive to sediment include SPEXP, SPCON, CH_COV1, CH_COV2, and the USLE_P. The NSE, R^2 and PBIAS for streamflow calibration were found to be 0.66, 0.69 and 10.3, respectively. The p-factor and r-factor were found to be 0.70 and 0.67, respectively and represented 70% of observed data bracketed within 95 PPU. The NSE, coefficient of determination and PBIAS were found to be 0.73, 0.75 and 7.20, respectively during validation of stream flow. The p-factor and r-factor for validation were 0.61 and 0.45. Spatial validation results of the model at gauging station 4CB04 indicated the model simulated stream flow correctly as indicated by an R^2 of 0.62. The annual average simulated sediment yield of 21.507 t/ha at the catchment outlet matched closely with the observed average annual value of 21.45 t/ha during sediment calibration.

The impact of contour farming, filter strips, terraces and grassed waterways on water and sediment yield were effectively simulated. Terraces were observed to be the most effective method in sediment reduction (80.7%) followed by grassed waterways (53.97%). Results showed that 46.04% of sediment would be reduced from the base annual average simulation value 21.507 t/ha. With 3 m width filter strips implemented in the catchment, 46.04% of sediment would be reduced at the catchment outlet. The effectiveness of sediment reduction was observed to increase with filter strip widths. However, filter strips beyond 15 m did not greatly impact on sediment yield reduction. Implementation of contour farming resulted to a decline of sediment yield at the catchment outlet by 35.81%. Contour farming and terraces resulted to a decline of stream flow by 16.40% and 30.25% while the baseflow increased by 3.38% and 8.38%, respectively. The total water yield decreased by 0.55% and 0.40% for contour farming and terraces, respectively. Grassed waterways and filter strips had no quantifiable impact on the total water yield from the catchment. Structural conservation measures were observed to be the most effective in reducing sediment yield. The Provisioning and regulating services of the ecosystem services are thus greatly influenced by soil and water conservation method applied.

Integration of different conservation indicated improvement of ecosystem service through regulation of sediment and water yield. Combination of grassed waterways, terraces and 3-meter filter strips reduced sediment yield by 90.42%. Grassed waterways combined with terraces reduced sediments yield by 88.72% while a 3-meter filter strips and grassed waterways reduced sediments by 74.55% from the baseline scenario value. Filters strips and terraces enhanced reduction in sediment yield by 84.92%. Contour farming and grassed waterways implemented together resulted in 70% reduction in sediment yield while contour farming implemented together with filter strips (3 meters) reduced sediment yield by 62.80%. It was also noted that filter strips and grassed waterways had no significant impact on water yield whether implemented individually or integrated with other methods. This study concludes that soil and water conservation methods can effectively be used to improve and restore the ecosystem processes and hence services provided.

5.2 Recommendations

To reduce model uncertainty and improve plausibility of results in sediments calibration, long term continuous spatial data should be collected. More research need to be conducted to expand the approach used in modelling soil and water conservation methods to include spatial distribution of such measures in the catchment. Further studies should be conducted in different regions and catchment/sub-catchment discretization to validate the results form experimental data and SWAT modelling results. The calibrated and validated model could be used to conduct other research on the impacts of land use change and climate variability of hydrological processes in the catchment.

A cost-benefit analysis should be conducted to establish the feasibility of implementing different soil and water conservation methods in a given area. Stakeholder involvement, capacity building and use of incentives e.g. through payments for ecosystem services (PES) programs should be should be encouraged to enhance adoption rate of these conservation methods. Institutions that manage water resources, especially the riparian areas should be strengthened to effectively execute their mandate in accordance with the law and provided guidelines.

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